JPL PUBLICATION 80-65

Analysis of Air Quality Management with Emphasis on Transportation Sources

Thomas D. English
Edward Divita
Lester Lees

October 15, 1980

Prepared for
U.S. Department of Transportation
Through an agreement with
National Aeronautics and Space Administration
by
Jet Propulsion Laboratory
California Institute of Technology
Pasadena, California

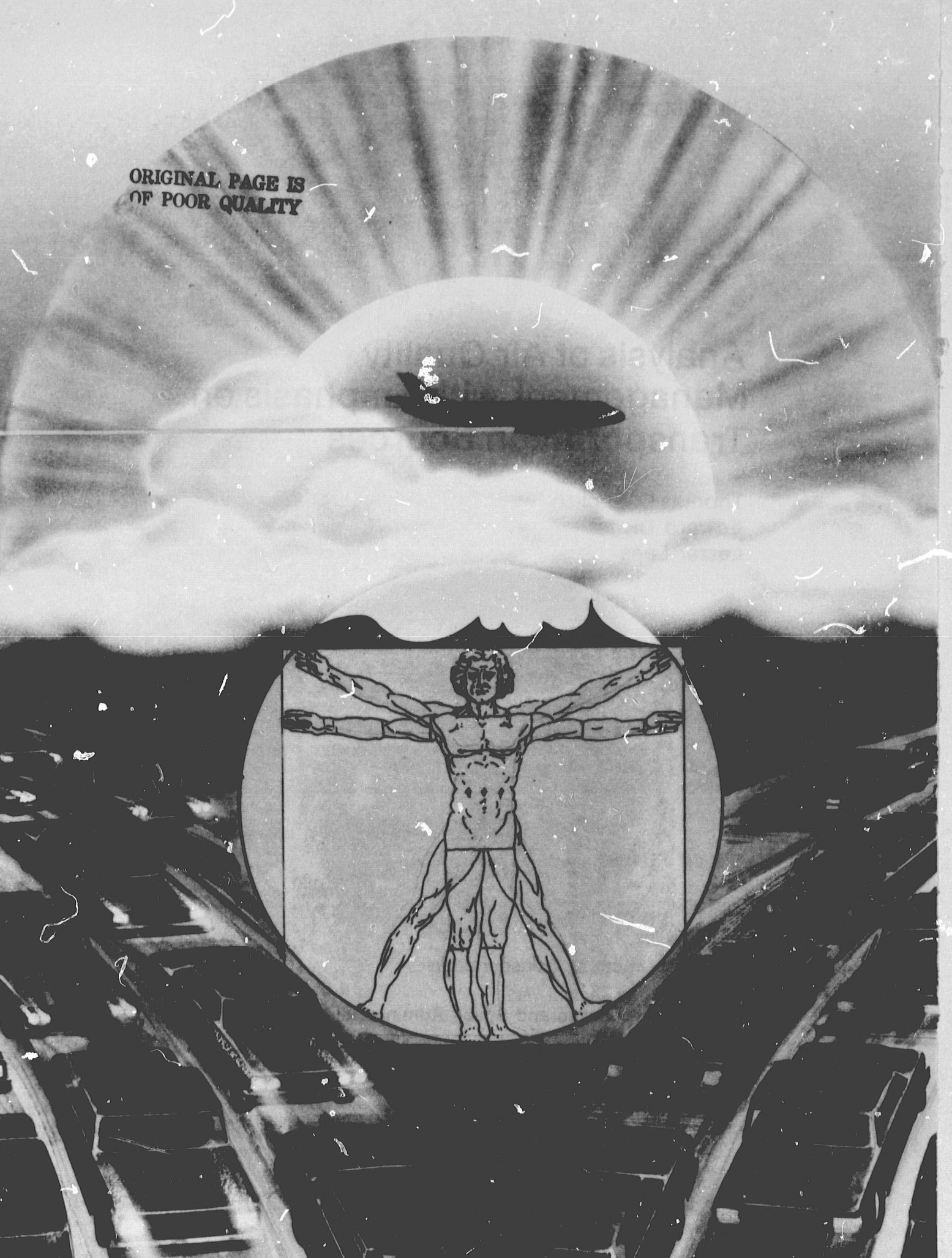
JPL PUBLICATION 80-65

Analysis of Air Quality Management with Emphasis on Transportation Sources

Thomas D. English
Edward Divita
Lester Lees

October 15, 1980

Prepared for
U.S. Department of Transportation
Through an agreement with
National Aeronautics and Space Administration
by
Jet Propulsion Laboratory
California Institute of Technology
Pasadena, California



FOREWORD

This report summarizes a portion of the results of a research project conducted by the Jet Propulsion Laboratory for the U.S. Department of Transportation. The purpose of this report was to characterize and assess transportation emissions inventories in terms of their impact on the air quality management process. The assessment included examining uncertainties associated with other sources of air pollutants, meteorological influences, atmospheric chemistry, air quality simulation models and air pollutant measurements.

The study was performed by personnel from the Jet Propulsion Laboratory and the California Institute of Technology. The duration of the study was approximately nine months, and involved about a man-year of effort. The work was funded under Contract NAS7-100. The Project Coordinator was Dr. Fredrick Marmo, Transportation Systems Center, U.S. Department of Transportation.

ACKNOWLEDGEMENTS

The authors are grateful for the time and effort provided by personnel from the following organizations:

American Lung Association
Arizona Department of Health Services
Arizona Department of Transportation
Caltrans
California Air Resources Board
Caltech's Environmental Quality Laboratory
Coalition for Clean Air
Colorado Air Pollution Control Division
Colorado Department of Highways
Denver Regional Council of Governments
National Commission on Air Quality
Sierra Club
South Coast Air Quality Management District
Southern California Association of Governments
United States Conference of Mayors
United States Department of Transportation
United States Environmental Protection Agency

We thank Stephen McReynolds for his assistance in preparing the section on transportation networks, Richard Blackwell for his preparation of the hourly maps of reactive hydrocarbon inventory, Greg McRae for several drawings from his work, and David Norris for initiating this project. In addition, we are thankful for the patience and skill of our secretary, Anne Shanahan.

EXECUTIVE SUMMARY

Based on this study of the tools currently available to air quality managers, we conclude that some observations on uncertainties in emissions, uncertainties in air quality models, and management air quality standards are important enough to warrant special attention.

1. Uncertainties in Emissions

Several air quality simulation models exist that are capable in principle of predicting the impact of transportation systems on ambient air quality. In practice, their utility is limited by the fact that uncertainties in pollutant emissions inventories are uncertain. For example, an estimate of uncertainty of ±60% has been made for reactive HC in the South Coast Air Quality Management District (SCAQMD). However, this estimate is currently not verifiable by the SCAQMD.

Based on both factory and surveillance tests, motor vehicle emissions of CO, HC and NO_x have substantially decreased during the last decade. However, data taken by the California Air Resources Board (CARB) indicate low correlation between emissions and mileage. This report discusses the reasons for the large variability in data, and shows that an Inspection and Maintenance program can cause a large decrease in this variability.

The use of Gaussian statistics to analyze auto exhaust emission data is misleading in cases where the standard deviation is greater than the mean.

In order to obtain meaningful "reasonable further progress" (RFP) trends, it may be necessary for the Congress to modify the Clean Air Act to require that both annual regional emissions and their uncertainties be estimated. If Congress imposes such a requirement, the funds which are required to determine these uncertainties could be authorized by the Congress or could be raised by either initiating or increasing emission fees.

2. Uncertainties in Air Quality Models

Air quality simulation models are subject to uncertainties in meteorological parameters, emissions inventories, rates of chemical reactions of the pollutants, measurements of ambient air quality, and the mathematical model itself. For example, the uncertainty in chemical kinetics is about ±50%. The uncertainties in emissions inventories may be comparable. Presently the combined effect of these uncertainties is unknown. A Monte Carlo method for determining the combined effect of the various uncertainties is described.

The credibility of photochemical air quality models, such as the empirical kinetic modeling approach (EKMA) is severely limited by the accuracy and precision of present measurement instruments for measuring NMHC and NO_x.

The prediction of the impact of transportation systems on ambient air quality is a statistical problem. At present the effect of changes in emissions from mobile sources is masked by the internal errors or uncertainties in the air quality simulation models. Since these uncertainties are large, only substantial reductions in RHC and NO_x emissions can lead to reliable improvement in oxidant ambient air quality.

3. Management Air Quality Standards

The present Federal ambient air quality standard for ozone is based on human health effect. The standard is expressed in terms of a maximum hourly average concentration not to be exceeded more than one day per year. Hence, present ozone air quality planning is based on a rare event. In the South Coast Air Quality Management District this standard is violated about 200 days per year, often for many hours a day. In other words, such violations are by no means rare events. This assessment of air quality simulation models shows that the models are poorly equipped to predict rare events because of the large uncertainties in the data base.

We recommend that serious consideration be given by the EPA to replacing the present procedure for compliance with the ozone amibient air quality standard with an approach which is statistically robust. For example, a "management standards" approach could be used which would set up milestones on the way to clean air in terms of hours per year that the Federal standard is exceeded. In those AQMD's that presently violate the Federal standard by a large margin, a reduction of 50% in the number of hours per year that the Federal level is exceeded might be mandated for the period 1979-1983. An additional reduction of 75% in the remaining number of hours per year that the Federal level is exceeded might be mandated for the period 1983-1987. This approach would permit statistically sound feedback to air quality managers responsible for planning the required reductions in emissions of specific pollutants.

CONTENTS

Figures

Tables

ABSTRACT

The current environment and practices of air quality management were examined for three regions: Denver, Phoenix and the South Coast Air Basin of California. These regions were chosen because the majority of their air pollution emissions are related to mobile sources. The report characterizes and assesses the impact of auto exhaust on the air quality management process. An examination of the uncertainties in air pollutant measurements, emission inventories, meteorological parameters, atmospheric chemistry and air quality simulation models is performed. The implications of these uncertainties to current air quality management practices is discussed. A set of corrective actions are recommended to reduce these uncertainties.

"...the truth shall make you free."

from Caltech logo
and
John 8:32

SECTION 1

INTRODUCTION

This report characterizes transportation sources of air pollutants, and assesses their impact on the air quality management process. This characterization examines uncertainties associated with the emission sources of air pollutants, meteorological influences, atmospheric chemistry, and air quality simulation models. This study includes an examination of the use of air quality simulation models by three cities which have predominantly transportation-related sources of air pollutants - Phoenix, Denver and Los Angeles. In summary, this report examines both the fundamental information base and the analytical tools which are used to make air quality management decisions.

In Section 1 of this report, a description of the post 1977 amendments to the Clean Air Act, institutional environment and its influence on air quality management is discussed. In addition, the approach that was used in this study is presented. In Section 2, the techniques for estimating the emissions for mobile sources for a given air basin are described. This section includes a discussion of exhaust standards, testing procedures, and transportation networks. In Section 3, a discussion of stationary source emission factors, and emission inventory systems is presented. Section 4 examines a spectrum of air quality simulation models which range from very simple to extremely complex. Estimates of the uncertainties of the models are made. Section 5 presents information on both the accuracy of instruments for measuring ambient concentrations of NO_x, NMHC AND O_3, and the implications this has on decision making. Air quality trends are also discussed. Section 6 summarizes the report and contains our findings, conclusions and recommendations.

1.1 STUDY ENVIRONMENT

The Clean Air Act Amendments of 1977 require the states to submit plans which show that the regions within the state can attain the primary ambient air quality standards by 1982. An extension to December 31, 1987, may be obtained if the district cannot meet the National Primary Ambient Air Quality standards for photochemical oxidents and/or carbon monoxide by 1982. (Section 1.7.2.A.2 of CAA). It should be noted that the assumption is made by the Congress that it is possible for all areas of the country to reach the National Ambient Air Quality Standards by either 1982 or 1987. In order to insure that the various districts and states make all reasonable attempts to reach these standards, the Clean Air Act Amendments of 1977 have put teeth into the Clean Air Act. That is, there are very severe sanctions that apply to the districts that do not attain the Primary Ambient Air Quality Standards by these dates. Section 1.1.0.A.2.1 provides that after June 30, 1979, no major stationary sources shall be constructed in areas for which the State Implemetation Plan has not been approved.

Other discretionary sanctions to the Administrator of EPA may also be invoked. The South Coast Air Quality Management Plan states that "Failure to implement these measures could result in Federal enforcement and impositions of sanctions against the funding and approval of all Federally funded projects, particularly transportation and waste water treatment projects." (Ref. 1-1.) Because these sanctions could amount to billions of dollars in some areas, they are taken very seriously by both the Air Quality Managment Districts and the states. This has resulted in the creation of comprehensive air quality management plans for many districts. For some districts, this is the first time they have ever had a plan that shows how they could attain the Primary Ambient Air Quality Standards.

1.1.1 State Implementation Plans (SIP)

Since the Clean Air Act was amended in August of 1977, the deadline of June 31, 1979, for State Implementation approval presented a formidable schedule to the three areas examined in this study, Phoenix, Denver, and the south coast air basin of California. The schedule was made particularly difficult by the requirement of Section 121 which states ". . . the state shall provide a satisfactory process of consultation with general purpose local governments, designated organizations of elected officials of local governments . . ." In the case of the Los Angeles area, this required that a draft of the Air Quality Management Plan be circulated for public review in October 1978 in order to allow revisions to be made before the Plan was submitted to the California Air Resources Board in January 1979. Hence, very little time was left for negotiations between the South Coast Air Quality Management District and the Southern California Association of Governments and the California Air Resources Board in order to finalize the air quality management plan as it would appear in the State Implementation Plan.

The complexity of the political process requiring local participation is indicated in the following: "SCAG is a voluntary regional organization composed of six Southern California counties--Imperial, Los Angeles, Orange, Riverside, San Bernardino, and Ventura--and approximately 130 member cities within these counties" (Ref. 1-2.).

Congress recognized many of the uncertainties in air quality management by requiring that annual reports be made which estimate progress toward achieving the national primary ambient air quality standards. It was thought that progress and problems could be identified on a yearly basis, and hence plans changed in order to attain the air quality according to a timetable. The Clean Air Act defines reasonable further progress as ". . . annual incremental reductions in emissions of the applicable air pollutant (including substantial reductions in the early years following approval or promulgation of planned provisions . . . and regular reductions thereafter which are sufficient in the judgment of the administrator, to provide for attainment of the applicable National Ambient Air Quality Standard by the date required in Section 172(a)."

According to a draft of the annual report in the South Coast Air Quality Management Plan dated April 11, 1980, p. viii, "the EPA recognized that some measures cannot result in immediate reductions and will accept a lag in decreases at the straight line rate for the first one or two years after 1979. Figure 1-1 shows an RFP schedule acceptable to EPA. It is important to note that RFP is measured by annual emissions reductions, not air quality."

1.1.2 EPA/DOT Interface

Transportation systems have a large impact on the quality of the environment. One of the major considerations when planning a new system is the effect on the quality of the air. This necessitates a close coordination of the requirements of both the Department of Transportation and the Environmental Protection Agency. Pursuant to the President's request, a Memorandum of Understanding was approved by the DOT and EPA in June 1978. It states that:

> In order to effectively achieve the objectives of the 1977 Clean Air Act Amendments, the DOT and Environmental Protection Agency agree that the transportation-related air quality planning requirements of EPA will be integrated with the transportation planning process administered by the DOT. Closer integration of the planning requirements of DOT and EPA will ensure the timely consideration of air quality concerns and will reduce potentially duplicative, over-lapping and inconsistent activities at the state and local level.

This mandates a significant DOT involvement in transportation/air quality impact assessment.

A model which could accurately predict air transport and transformation phenomena would be an invaluable tool for understanding the effect of a proposed transportation change. In fact, such a model is necessary to satisfy the analysis requirements of the various highway transportation acts and clean air acts which relate to the legal responsibilities of the Secretary of Transportation to curb the impact of transportation-generated air pollution on the environment.

The adequacy of present air pollutant emission and air dispersion models for analyzing the mesoscale and regional air pollution impact of transportation systems has been questioned by the Department of Transportation. Data from such models are required to perform the analyses mandated by the various highway transportation acts and the Clean Air Act Amendments (CAAA) of 1977. In particular, under the CAAA, certain states must include such analyses in their 1982 State Implementation Plans (SIP's).

The EPA recognizes the present state of air quality models and has initiated a comprehensive program to validate selected grid and/or trajectory air quality simulation models (AQSM's) for five cities: Denver, San Francisco, Los Angeles, Tulsa, and St. Louis. The program is to provide user agencies with a set of validated AQSM's which have been jointly (EPA/DOT) approved for preparing their 1982 SIP's.

The necessity for joint approval is indicated by the memorandum of understanding between DOT and EPA, June 1978, which "established a DOT/EPA coordination process to reduce potentially overlapping and inconsistent activities and provide for a single Federal decision on local clean air proposals".

To provide the DOT with information relevant to its responsibilities, this work characterizes the transportation sources' inventories (TSI's) and assesses their impact on the air quality management process.

1.1.3 National Commission on Air Quality

The 1977 Amendments to the Clean Air Act were so extensive that the Congress decided to form a National Commission on Air Quality to examine the pros and cons of meeting or not meeting the requirements of the 1977 Amendments. The charter of the National Commission on Air Quality is spelled out in detail in Section 323 of the Clean Air Act.

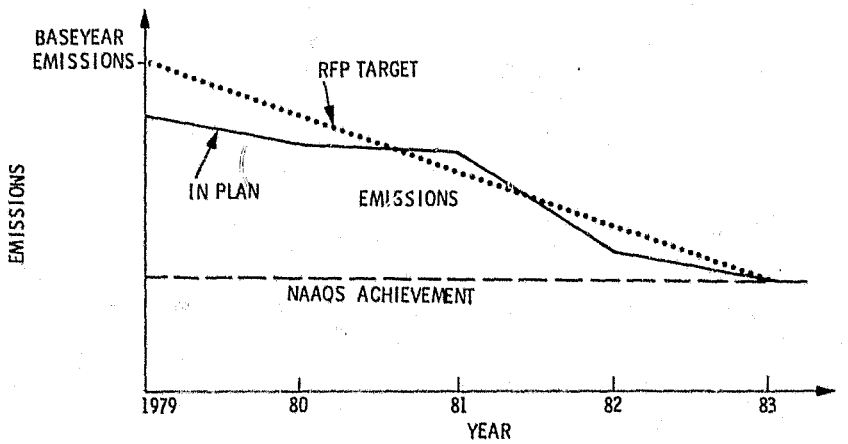


Fig. 1-1. Reasonable Further Progress Schedule

The work plan for the Commission has been published in the Federal Register July 6, 1979, page 39636 (Ref. 1-3.). The principal components of the plan include the following:

I. Analysis of National Ambient Air Quality Standards (NAAQS)
II. Review and analysis of policy for areas attaining standards
III. Review and analysis of policy for nonattainment areas
IV. Review of vehicle emissions standards
V. Cost and benefits associated with air pollution control
VI. Impact of air pollution abatement activities on selected industries
VII. Review and analysis of institutional relationships and research programs
VIII. Review and analysis of alternative approaches to air pollution control

The Commission has been holding a set of conferences in various areas of air quality management. Of particular interest to this study was the workshop held on Photochemical Modelling. This workshop will be described in some detail later in this report.

1.2 APPROACH AND SCOPE

In order to characterize and assess the requirements of transportation air pollutant source inventories used in air quality simulation models, this study emphasizes an analysis of the impact of technical uncertainties on air quality management. A literature review of emissions inventories and air quality simulation models (AQSMs) was performed. The pollutants included in this review were NO_x, O_3, HC's, SO_2, TSP, CO and Pb. A background on the Federal view of current issues and techniques of air quality management was obtained through site visits to the Environmental Protection Agency's centers at Research Triangle Park, N.C.; Durham, N.C.; Washington, D.C.; and San Francisco, CA. In addition, a visit was made to the headquarters of the National Commission on Air Quality. During these visits, discussions took place with air quality modelers, regulators and planners concerning the detailed manner in which AQSMs are used. After these site visits, a series of discussions took place between JPL and Environmental Quality Laboratory (EQL) personnel concerning the sensitivity of the outputs of AQSMs to the transportation source inventory (TSI) inputs, stationary source inputs, and other pertinent factors.

One study goal was to determine whether significant gaps or inadequacies existed in present approaches to transportation source inventories, stationary source inventories, meteorological factors and air quality simulation models. Another goal was to recommend corrective actions which would mitigate the effects of the gaps. One of the best sources of knowledge of current issues and gaps in air quality management are the state and local air quality managers. In order to obtain a sampling of their views of air quality issues we talked with a variety of state and local organizations which are active in air quality management. In particular, we focused our attention on personnel with responsibility for the air quality management regions surrounding Phoenix, Arizona; Denver, Colorado; and Los Angeles (South Coast Air Quality Management District).

These regions were chosen because most of their pollution sources are related to mobile sources (Refs. 1-1, 1-2, 1-3 and 1-4). The percentage of total emissions due to mobile sources is shown in Fig. 1-2 for these regions. Data are presented for hydrocarbons (HC), oxides of nitrogen (NO_x), carbon monoxide (CO), total suspended particulate (TSP), and sulfur oxides (SO_x). Over 90% of the CO originates from mobile sources for these regions. For the Denver region, 85% of the HC and 76% of the TSP are from mobile sources.

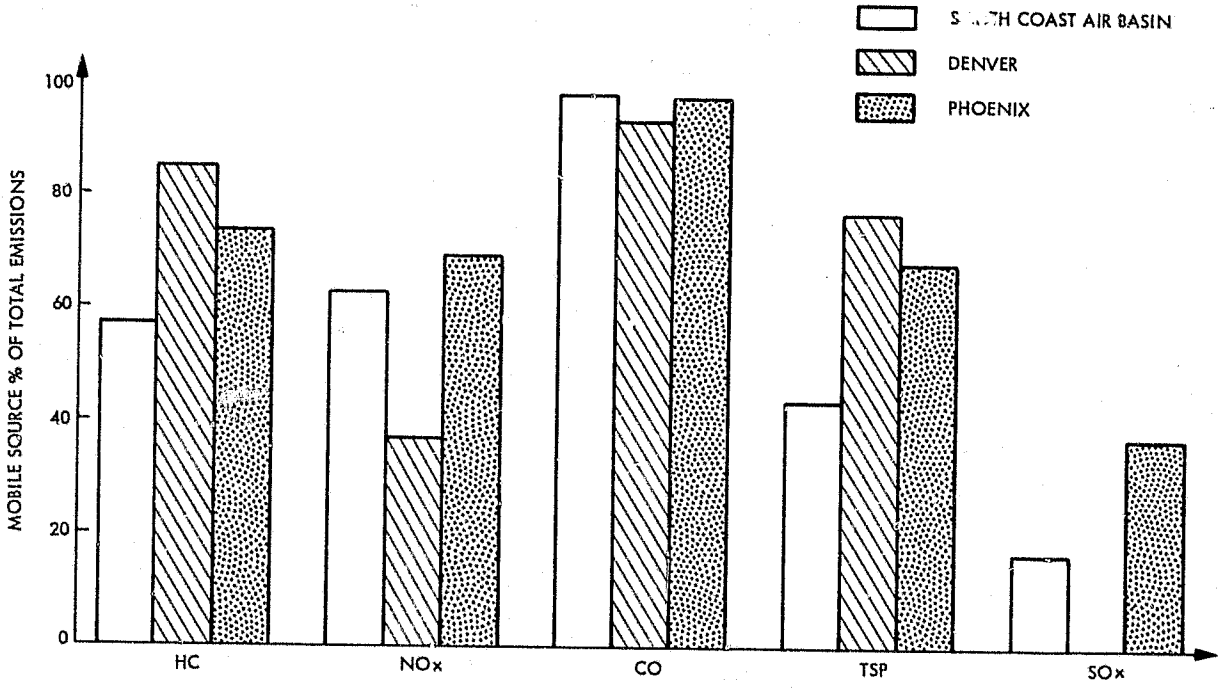


Fig. 1-2. Mobile Sources Percent of Total Emissions Based on State Implementation Plans for South Coast Air Basin, Denver, and Phoenix

We visited with two groups in Arizona, the Department of Health Services, and the Arizona Department of Transportation. The Arizona Department of Health Services includes an Environmental Health Services Division, which has a bureau of air quality control. The site visit to Phoenix, Arizona, was enhanced by the Clean Air Conference that the United States Conference of Mayors held in that city. In Denver, Colorado, we visited with representatives from the Denver Regional Council of Governments, the Colorado Department of Health, and the Colorado Division of Highways. In Colorado, the Air Pollution Control Division is within the Colorado Department of Health.

Our major interactions during this study took place in the California area. Visits were made to the California Air Resources Board to get the state view of the State Implementation Plan process. An extensive set of interactions took place between our staff and the staffs of both the Southern California Association of Governments (SCAG) and the Southern California Air Quality Management District (SCAQMD). Many of the ideas that are expressed in this report came from the staffs of the agencies cited above. However, some are solely the responsibility of the authors.

It should be noted that the Clean Air Conference held by the United States Conference of Mayors in Los Angeles, August 23-24, 1979, provided useful background information for this study. Some of the material from both the L.A. and the Phoenix conferences will be described later in the report (Refs. 1-4, 1-5, 1-6).

1.3 BACKGROUND ON AIR QUALITY SIMULATION MODELS

In general, Air Quality Simulation Models (AQSM's) are used to relate the emissions of air pollutants from a variety of sources to the ground level concentration of each air pollutant. These models range in complexity from the "back of an envelope calculation," to extremely elaborate computer simulation models that run at a speed approximately equal to the speed at which the air pollution reactions take place in the atmosphere. In other words, the simulation of one day of air quality might require several hours of computer running time. The general components of an air quality simulation model are shown in Fig. 1-3. The more complex air quality simulation models attempt to account for the sources of air pollutants, their transportation and dispersion by the winds, the chemical transformations that take place, and the removal mechanisms that affect them.

The sources include both man-made sources of air pollutants, such as stationary power plants or mobile emissions from automobiles, and geogenic sources, i.e., sources directly from the earth. An important source of pollutants are those that are transported into an air quality region from outside that region. The meteorological component in Fig. 1-3 includes physical parameters such as temperature, the height of an inversion layer, the wind field, the turbulence associated with the wind, surface topography and surface roughness. Cloud cover affects the amount of radiation coming from the sun; hence it influences both the temperature distribution and the chemical reactions.

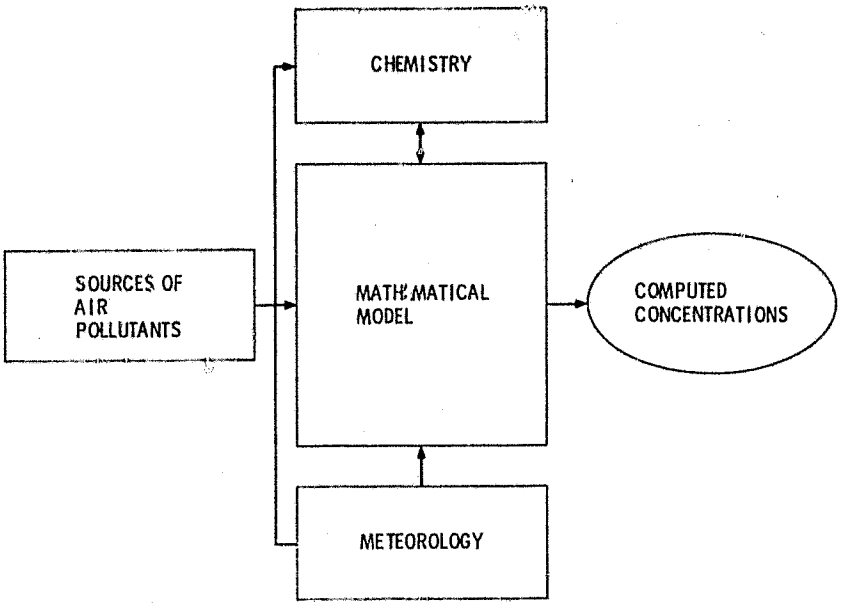


Fig. 1-3. Major Components of Air Quality Simulation Models

The box labeled chemistry at the top of the diagram in Fig. 1-3, would, for example, include:

(1) Photochemical reactions which cause oxides of nitrogen and hydrocarbons to react to produce ozone and other components of photochemical smog.

(2) Homogeneous processes involving gaseous reactions.

(3) Heterogeneous processes involving gases and particulates or aerosols.

(4) Removal processes surface deposition by contact, rainout, etc.

The sources of the air pollutants, the meteorology, and the chemistry are put into a mathematical model which is an analytical description of the interaction of these components. The output of the model is the concentration of air pollutants. These concentrations may be considered a constant over a region in the simple models or as a function of three spatial coordinates in the more complicated models. The above fairly complex type of model contrasts to the simple models used in rollback type calculations.

Warren B. Johnson, in Ref. 1-7, provides a summary of some of the reasons for air quality simulation models.

Questions such as the following must then be considered:

- What are the relative contributions of various current or planned source categories or individual sources to ground level concentrations?
- What will be the effect of a reduction in emissions of one of the reactive pollutants involved in producing a secondary pollutant?
- What air quality improvements can be achieved through indirect controls; i.e., changes in the source distribution in space and time?

> The only hope for answering such 'what if . . .?' questions lies in air quality simulation models which furnish a scientific basis for decisions leading to least-cost air quality improvement.

To assist in addressing these questions, the current usage patterns and the uncertainties associated with the following air quality simulation models are examined in this report.

(1) Rollback.

(2) Empirical Kinetic Modeling Approach (EKMA).

(3) Photochemical dispersion.

Each of these models requires a knowledge of the sources of pollutants in the area under study, i.e., each of these models requires an emissions inventory.

1.4 EMISSION INVENTORIES

Each city and region of the United States is characterized by an extremely complicated set of activities. These activities will vary from region to region and city to city. These activities are associated with the emissions of air pollutants. Hence to keep track of the emissions in a given area, a system is required which categorizes the pollutants emitted by species, location, rate of emission, and other pertinent characteristics. The structure of an emissions inventory is shown in Fig. 1-4. This inventory is based on the structure used in the National Emissions Data System (NEDS) (Ref. 1-8.)

The primary breakdown in the emissions sources is whether the sources are stationary or mobile. Stationary sources can be considered as point sources where large amounts of pollutants come out of a single smokestack or complex of smokestacks Area sources are referred to as either combustion area sources or noncombustion area sources. Typical area combustion sources are residences, commercial establishments, institutional establishments such as schools, hospitals, public buildings, and some small industrial establishments. Noncombustion area sources may emit hydrocarbons such as vapors from gasoline storage and handling by distributors and service stations, dry cleaners, painting, degreasing, and application of asphalt to roofing and pavement.

The National Emission Data System divides point stationary sources into two categories, internal combustion and external combustion. External combustion, as shown in Fig. 1-4, is subdivided into fuel combustion, industrial processes and solid waste disposal. Examples of fuel combustion sources include residential sources, electrical generation sources, industrial fuel sources, etc. Industrial processes include chemical manufacturing, primary metal manufacturing, secondary metal manufacturing, etc. The mobile sources are subdivided, as shown in Fig. 1-4, into land vehicles, ships and aircraft. The land vehicles are broken down into gasoline and diesel fueled vehicles. Included under diesel vehicles are trains, off-highway vehicles, light duty vehicles and heavy duty vehicles.

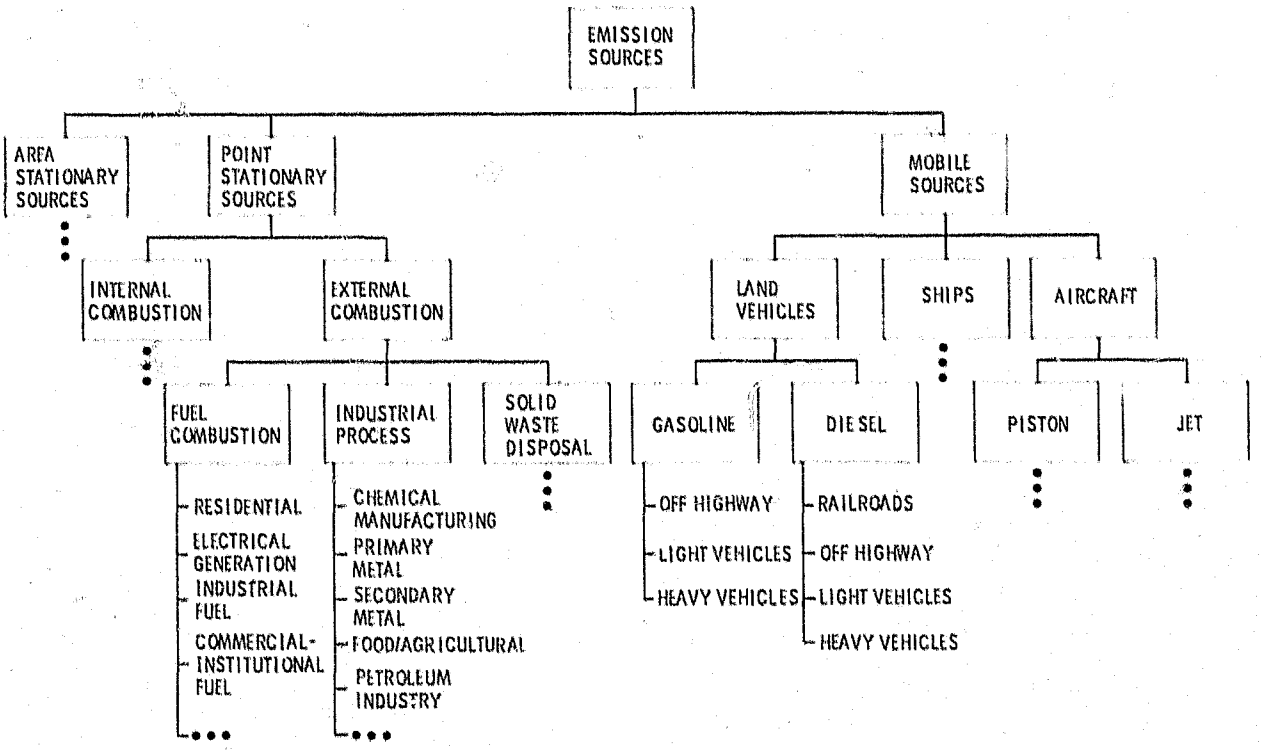


Fig. 1-4. Work Breakdown Structure for Emission Sources

This report focuses on light duty vehicles. McCray (Ref. 1-9) makes the following comments concerning emissions inventories: "Without a doubt, the most important input to any air-shed model (photochemical dispersion model) is a comprehensive, detailed and accurate emissions inventory. What is often neglected is that regardless of the approach used to relate emissions to ambient air quality, it is also impossible to design an efficient oxident control strategy without an adequate inventory. When constructing an inventory, it is necessary to assemble the source emissions data at a level of accuracy consistent with the required spatial, temporal and chemical resolution of the problem."

"By structuring the inventory in the manner shown, it is possible to vary, for example, the emissions from mobile sources and in fact from particular vehicle classes without altering the remainder of the inventory. While this structure is not required as part of the air quality calculation, it considerably simplifies the task of constructing control strategies from a list of alternative emission control tactics."

As is shown in Fig. 1-3, the entire air quality simulation modeling process cannot be any better than our knowledge of the sources of the air pollutants, that is, the emissions inventory. It should be noted that separate emission inventories are required for each category of pollutant under consideration, e.g., carbon monoxide, sulphur dioxide, nitrogen dioxide, and total suspended particulate. In the case of photochemical smog or ozone, the situation is much more complex, since emissions inventories for both the oxides of nitrogen and reactive hydrocarbons are required because these pollutants interact in the formation of photochemical smog. The chemistry of these interactions is discussed later in this report.

Now that the current environment for air quality management and the relative roles of the major institutions have been described, it is appropriate to examine the capabilities of the air quality manager in detail. In the next four sections the current status of emissions inventories for mobile and stationary sources is described, followed by a discussion of models and measurement capabilities.

REFERENCES

1-1 Air Quality Management Plan, South Coast Air Quality Management District and Southern California Association of Governments, El Monte, Calif., Jan. 1979, p. i-1.

1-2 "SCAG Growth Forecast Policy 1978," Southern California Association of Governments, Los Angeles, Calif., 1978.

1-3 "Air Waves," National Commission on Air Quality, Capitol Hill, Washington, D.C., Vol. 1, No. 1, Aug. 1979.

1-4 Air Quality Management Plan, South Coast Air Quality Management District and Southern California Association of Governments, El Monte, Calif., p. V-17, Jan. 1979.

1-5 SIP Revision - Model Analysis of Existing and Future Air Quality Levels in the Denver Area, Analysis Unit, Planning and Analysis Section, Air Pollution Control Division, Colorado Department of Health, p. B-425, December 8, 1978.

1-6 "Technical Support Documentation--Phoenix Air Quality Maintenance Plan for Carbon Monoxide and Photochemical Oxidants," Arizona Department of Transportation, Transportation Planning Division, B-78-1, March 1978.

1-7 Urban Air, Postconference Report, United States Conference of Mayors, August 23-24, 1979, Los Angeles, California

1-8 Urban Air, Preconference Handbook, United States Conference of Mayors, January 10-11, 1980, Phoenix, Arizona.

1-9 Johnson, Warren B., "Activities and Needs in Air Quality Simulation Models," 6th Hawaii International Conference on System Science, Special Sub-Conference on Regional and Urban Systems: Modelling Analysis and Decision Making, January 9-11, 1973, Honolulu, Hawaii.

SECTION 2

MOBILE SOURCES

This section discusses the emissions from mobile sources, emphasizing light duty vehicles. Data used in this section are from many tests following the Federal Test Procedures. Test results include those from both factory and surveillance testing. Idle and evaporative emissions are included. An analysis is performed to examine the influence of variability and uncertainties on emissions as a function of model year and odometer mileage. It compares the data from in-use cars obtained by EPA and California Air Resources Board. In particular, the data from "as received" cars and repaired cars are used to demonstrate the potential emissions reduction obtainable through Inspection and Maintenance. Finally, this section identifies characteristics of transportation network models.

Mobile sources of primary interest in this study include Light Duty Vehicles (LDV) because they are the predominant source of emissions of HC, CO and NO_x. Emissions of HC, CO and NO_x from LDV cause major increases in ambient air pollution throughout the United States. Cities such as Denver, Phoenix and Los Angeles are particularly impacted by the emissions from mobile sources. These cities are used as examples in the assessment of the emissions from LDV sources. In particular, test results obtained from California cars, high altitude cars (Denver), and other cars tested throughout the United States are used to examine uncertainties and variability in the test data. The study compares data from several data bases with the appropriate vehicle emission standards. In addition, the reader is appraised of the problems associated with interpreting and using the test results for projecting future emissions and planning strategies. The study reviews the surveillance testing currently underway. Even though the results point conclusively to substantial decreases in the emission levels in the newer LDV's, the uncertainty in the data is still large. This study does not examine the technical adequacy of emissions control systems; however, it does compare the test results on cars for model years 1970-1979. Newer model year data demonstrate that the use of catalyst on LDV's is an effective control for emissions. Other test results such as emission during idle testing and during evaporation tests are also discussed to show both the uncertainties and the importance of these emissions as major sources during the typical use of a vehicle.

2.1 MOBILE SOURCES WORK BREAKDOWN STRUCTURE

The mobile sources work breakdown structure presented in Fig. 1-4 and previously described in a general manner identifies all of the important mobile sources. In particular, the light duty vehicles under gasoline powered land vehicles are the dominant source of emissions. This fact is displayed in Fig. 2-1, which shows the projection of the total vehicles in use through 1985 (Ref. 2-1). Automobiles constitute more than 80% of the vehicles.

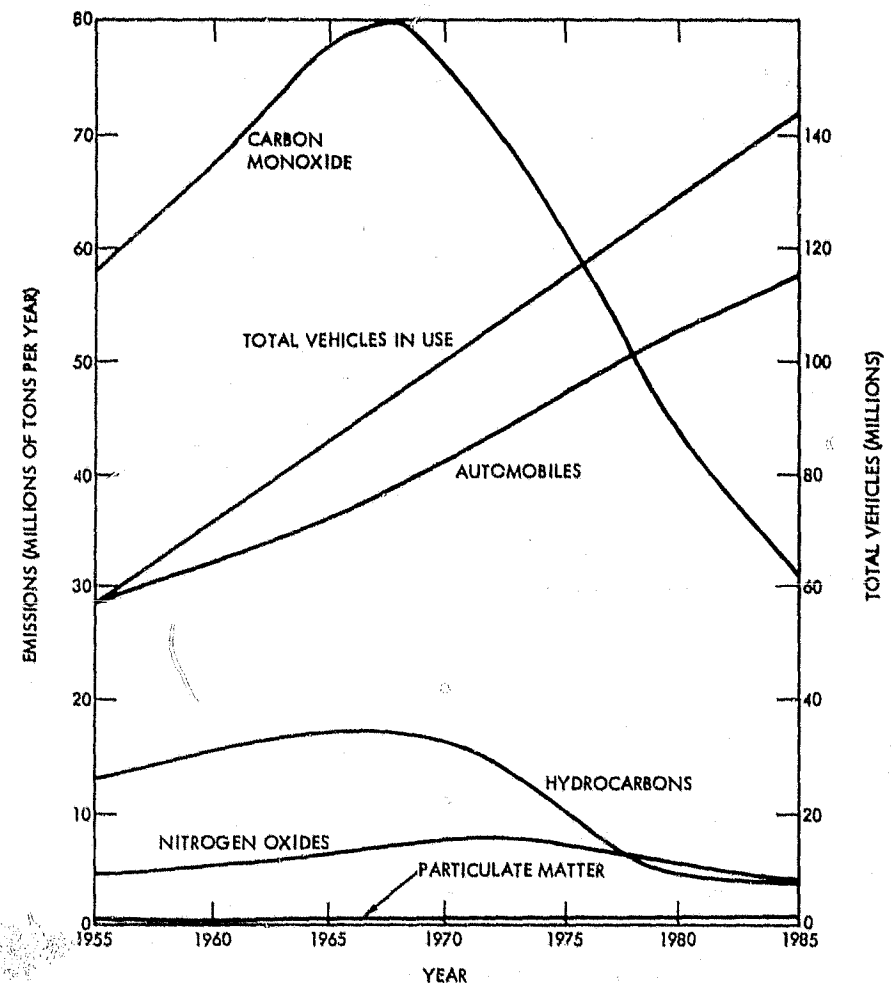


Fig. 2-1. Mobile Source Annual Emission Rates (Ref. 2-1)

The related emission of CO, NO_x and HC are shown also as a function of time to indicate the value of controlling the emissions from these vehicles. Note the changes that started in 1968 and the projected continuous decrease of these major constituents, even though the number of vehicles is increasing. This decreasing net emissions trend is substantiated by the detailed test data discussed in this section.

2.2 MOTOR VEHICLE PROJECTIONS

Recent projections for 1979 through 1990 (Ref. 2-2) based on current data from the EPA Emission Factor Program and results of the EPA Certification programs have been made to determine future vehicle emissions. The projections are made for U.S. as presented in Figs. 2-2 and 2-3, which show, respectively, the projected growth in total vehicle registrations and a projected light duty vehicle utilization index (average annual VMT per LDV divided by the corresponding value for 1972). These figures show an increasing trend through 1990. The assumptions related to projections in Fig. 2-2 are detailed in Ref. 2-2 but do not account for the past two years of increased inflation and major increases in gasoline prices.

2.3 LIGHT DUTY VEHICLE EMISSIONS

Motor vehicle Federal and California standards are tabulated through 1983 in Appendix A. A summary of how the Federal standard has changed since 1967 and as planned for 1983 is shown in Figure 2-4 for NO_x, HC, and CO. It is important to note that the overall decrease in the HC and CO standards has been about a factor of 20, while that for NO_x is less than a factor of 5.

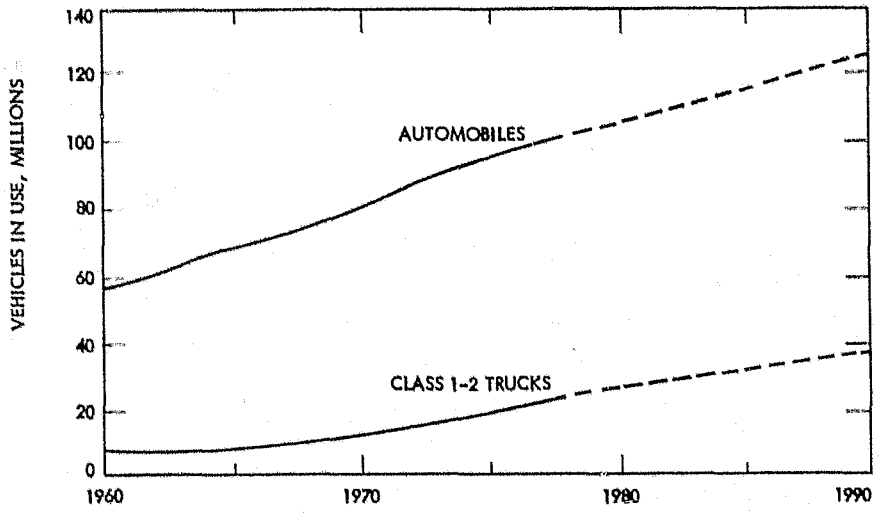


Fig. 2-2. Projected Growth in Total U.S. Motor Vehicle Registration (Ref. 2-2)

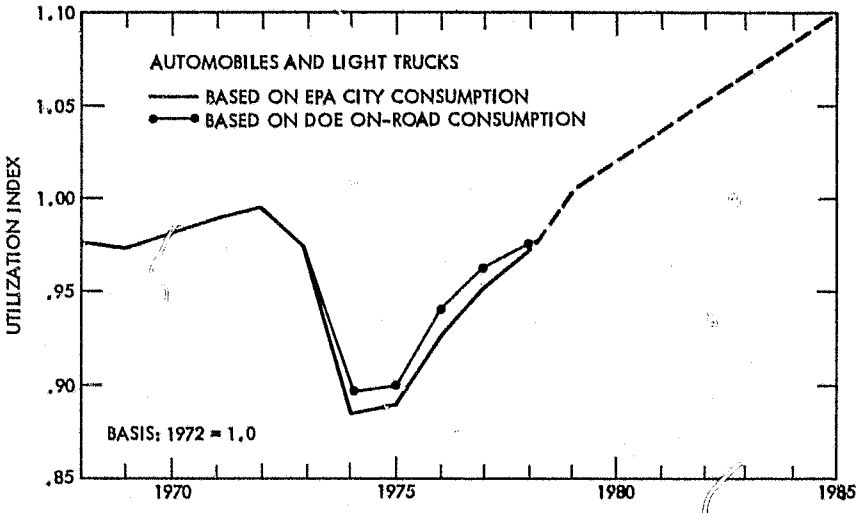


Fig. 2-3. Projected Relative Light Duty Vehicle Utilization Index (Ref. 2-2)

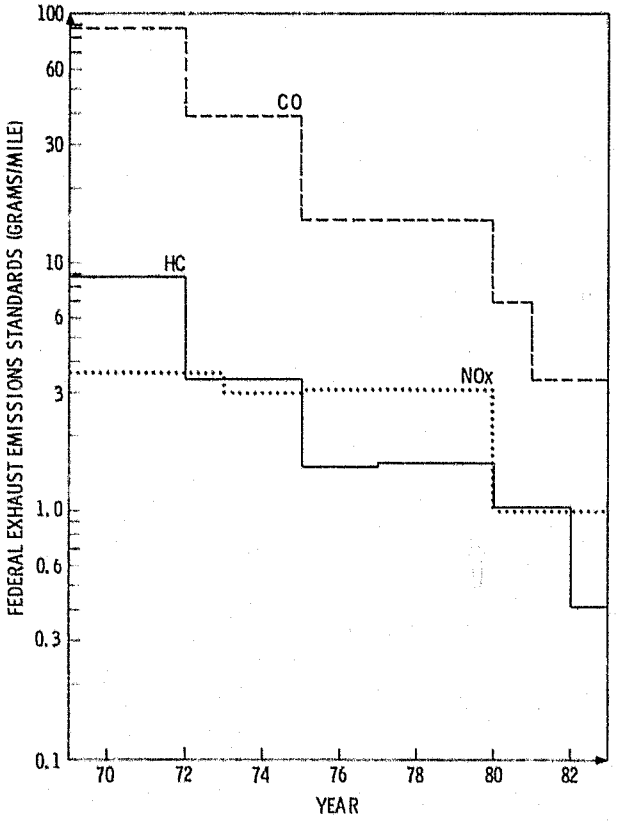


Fig. 2-4. Summary of Changes in the Federal LDV Standards Since 1967

The State of California has established exhaust emission standards which, in many cases, are more stringent than the Federal standards. This was done in an attempt to alleviate the serious photochemical smog problems which impact several of California's heavily populated air basins. For example, the South Coast Air Basin, which contains approximately 10 million people, exceeds the national primary ambient air quality standard for ozone about 200 days per year. The California standards from 1966 to 1983 are shown in Appendix A. Many tests have been performed to determine the "in use" performance of LDV emission control systems. Both factory and field test method and procedures are described in parts B.1 and B.2 of Appendix B of this report. The exhaust test data show considerably variability. In many cases the standard deviation of the data is approximately equal to or greater than the mean of the data. The relative contributions of vehicle, driver, ambient conditions and exhaust gas measurement instruments are described in part B.2.3. It is of interest to note, for example, that ambient conditions have a small influence on variability of HC and NOx emissions, whereas they are the largest contributor to CO variability.

The California Air Resources Board has performed a set of tests of the emissions of cars to determine both their "as received" exhaust characteristics, and their characteristics after repair according to a protocol. The data are examined for given model years as a function of odometer reading. The data are statistically summarized by determining both the best straight line fit of the data (exhaust vs mileage) and the correlation of the fit. The correlation coefficients found are less than or equal to 0.3. A detailed discussion of the CARB results is presented in part B.3 of Appendix B. The repair procedures produced substantial reductions in emissions. Figure 2-5 shows both the "as received" exhaust characteristics and the "after repair" characteristics. It should be noted that the average car does not meet the emission standard after approximately 20,000 miles, even after the car has been repaired.

The value of an inspection and maintenance (I/M) program is also shown in Fig. 2-5. If the I/M procedures are performed every 10,000 miles, then exhaust emissions are reduced to the "after repair" straight line. The assumption is made that the exhaust emissions then increase with mileage, with the same slope as to the "as received" cars. Hence, with an I/M program, the emissions follow the saw-toothed pattern shown in Fig. 2-5. The emissions reduction associated with I/M can be substantial. For example, the crossed-hatched area represents the emissions reductions obtainable with I/M between the car's 50,000th and 60,000th mile. In this example this emissions saving is about 45%.

The point demonstrated by the sawtoothed curve is that little deviation will occur per the average car if repaired systematically. In a quantitative assessment considering the CARB data and California standards the NO_x and CO standard might be met by the average 1975 through 1979 cars having an accumulation of 50,000 to 70,000 miles. Even with I/M, the HC emissions on the average 1975-1976 and 1977-1979 cars would not be within the standards but the improvement

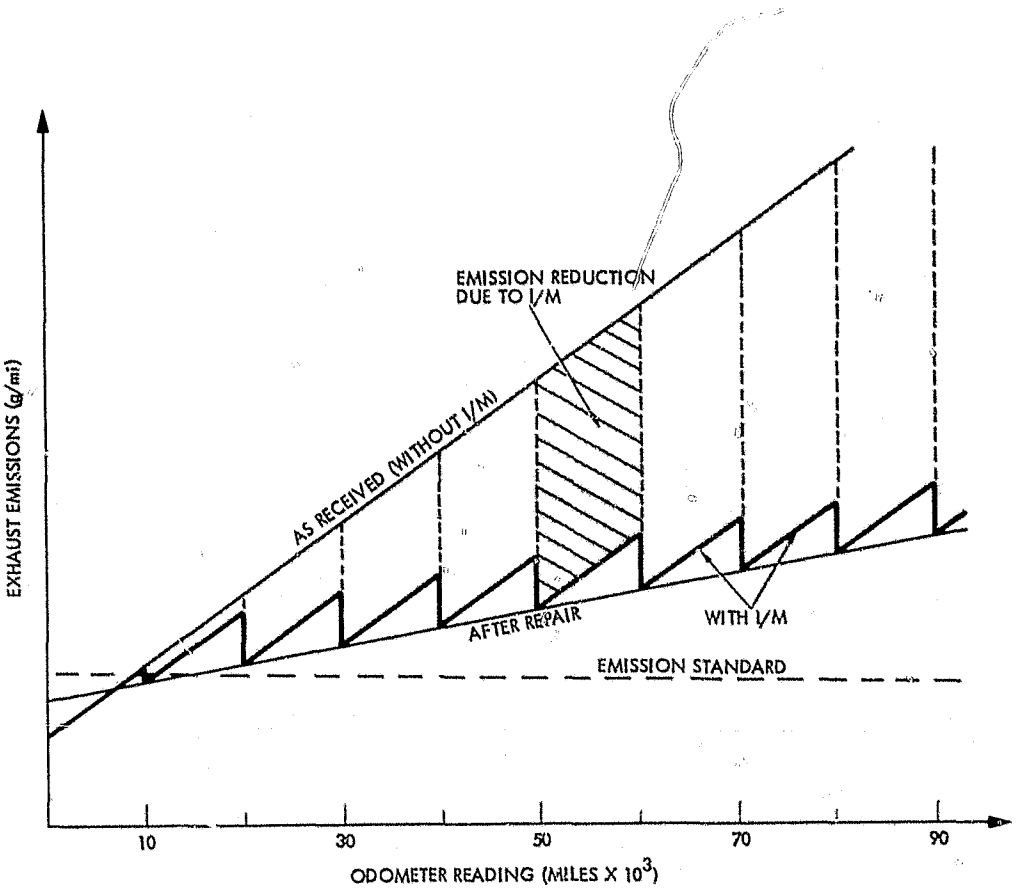


Fig. 2-5. Model of Potential Emission Reduction Due to I/M versus Odometer Reading

would be significant. For HC emissions the 1975-1976 and 1977-1979 average cars would exceed the standard by a factor of 2 at 70,000 miles.

This estimate is based on using AP-42 methodology and assuming every car in the U. S. will be built according to existing standards including those to go into effect for 1981 models. It further assumes that the average car will reach its 50,000-mile point before four years of age.

The improvement in California cars is shown in Figs. 2-6, 2-7 and 2-8, based on data presented in Ref. 2-5. These figures show the mean exhaust emissions as a function of model year. In the figures (see Fig. 2-6 for example), the number of cars in the sample as well as the 1σ deviation on the test results is shown. The characteristic trend in the data indicates that the newer model cars are emitting less pollutants. However, for some years, especially for CO emissions (see Fig. 2-7) the deviations associated with the newer cars are as large as those of earlier model years.

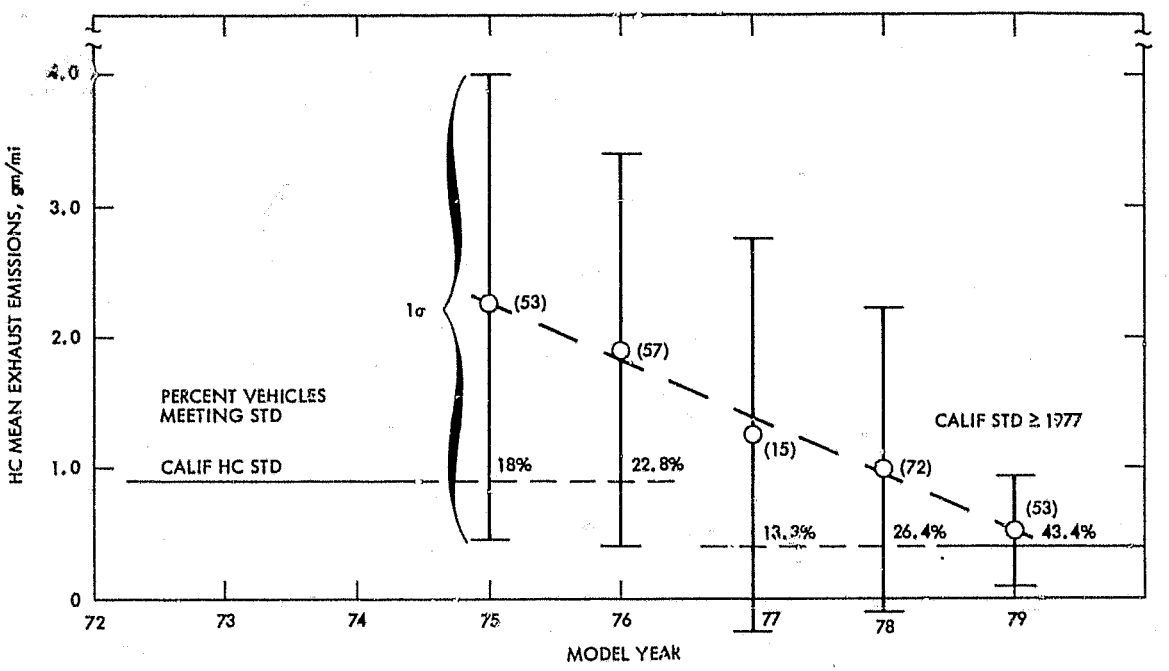


Fig. 2-6. HC Mean Emissions and Standard Deviations versus Model Year of California Cars

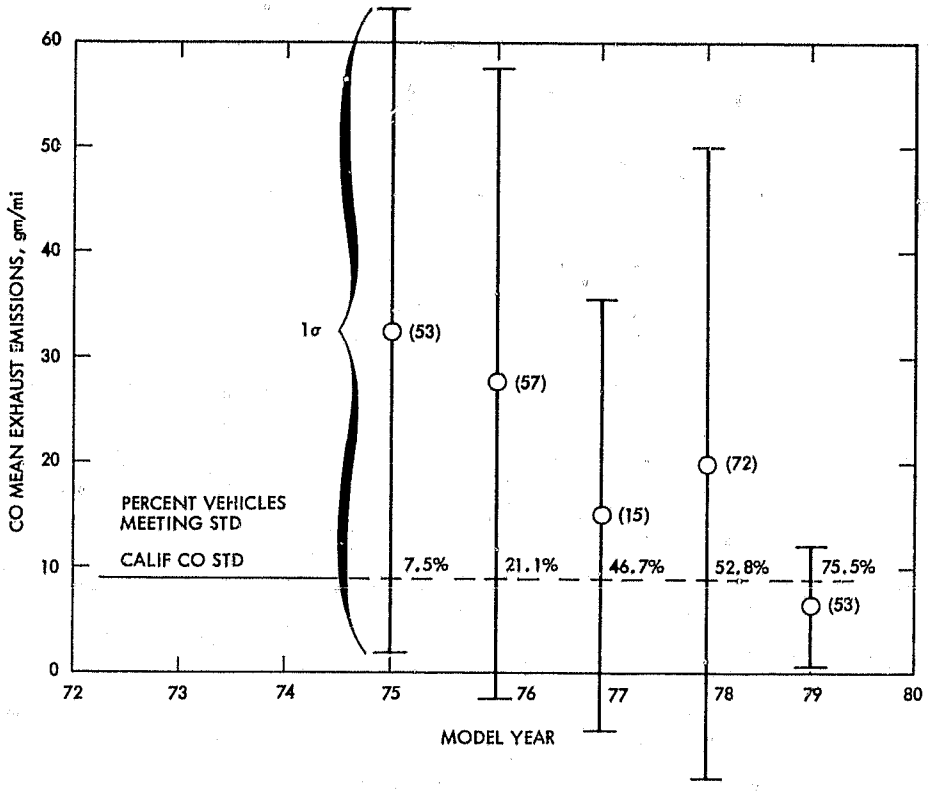


Fig. 2-7. CO Mean Emissions and Standard Deviations versus Model Year of California Cars

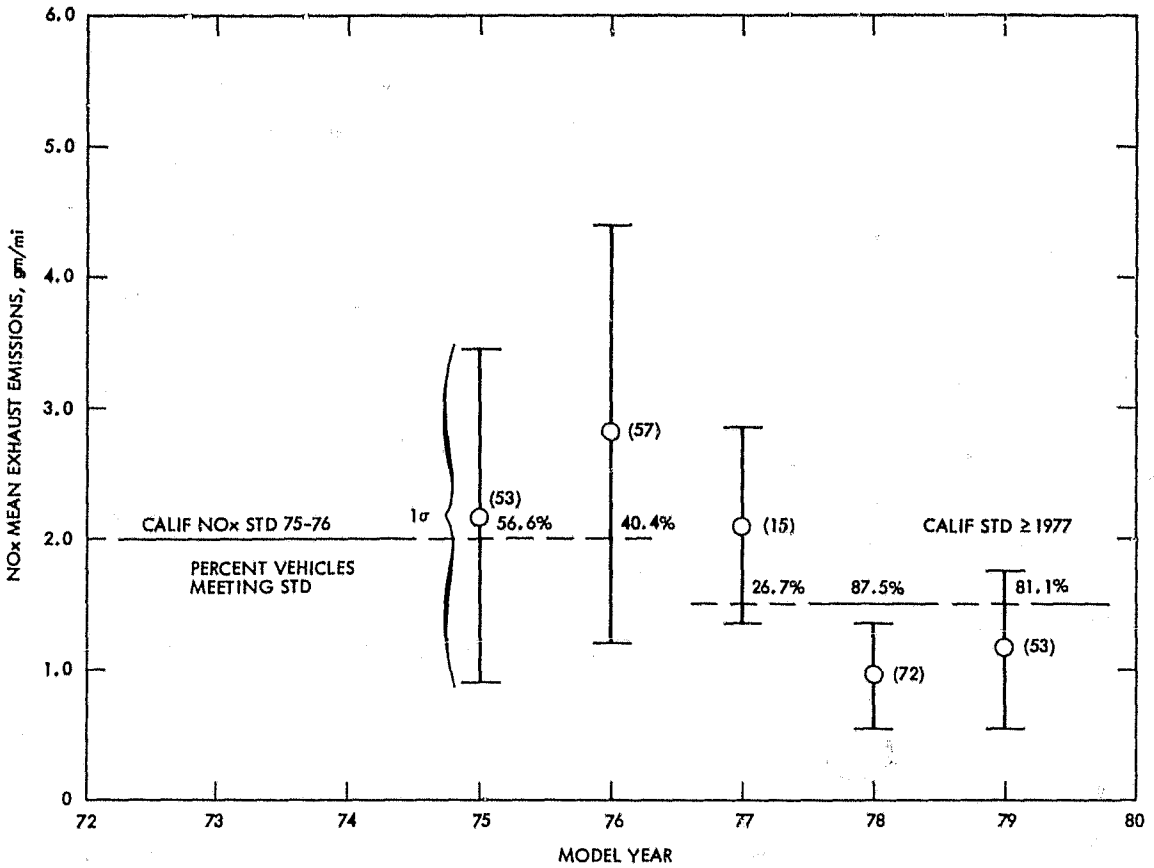


Fig. 2-8. NO_x Mean Emissions and Standard Deviations versus Model Year of California Cars

A further quantitative examination of surveillance test data as performed in Ref. 2-4 demonstrates the effect of restorative maintenance on LDV. The results further support the need for developing effective I/M programs for in-use LDV (and other vehicles). As an example, Figs. 2-9 and 2-10 show results that demonstrate the significant improvement in relatively new cars that have been tested as received and then tested after restorative maintenance. The results shown are based on tests of cars in service in Detroit, Chicago and Washington, D.C. In this case a 50 percent improvement in the number of cars that passed the Federal standards is shown.

More region-specific data were also obtained during in-use tests with Denver, Detroit and Los Angeles cars. The results presented in Figs. 2-11 and 2-12 show that except for Los Angeles the improvement after restorative maintenance even for new cars with <4000 miles is almost a factor of 2 between those that pass. The Los Angeles cars showed that of the remaining cars that failed almost 1/2 of those passed after restorative maintenance.

This large a percentage pass of Los Angeles cars before maintenance is indicative of the more stringent set of certification requirements that the vehicles must pass to qualify as a California car.

Based on these test results (Ref. 2-4), the percent of emission component malperformance is tabulated in Table 2-1 for comparison with the allowed maintenance of California cars according to the I/M procedures. In addition, the results of examining the percentage emission increases due to various maladjustments and disablements on the 1975 and 1976 vehicles are shown in Table 2-2. Typically, at least one increase per emission component is significant for a given type of malperformance. However, as noted in Table 2-1, using Los Angeles cars for comparison, the three contributions to malperformance occur with the carb/choke/exhaust heat control system (30%), ignition system (10%), and EGR system (7%). In this case, potentially minor repairs could result in significant improvement in Los Angeles cars.

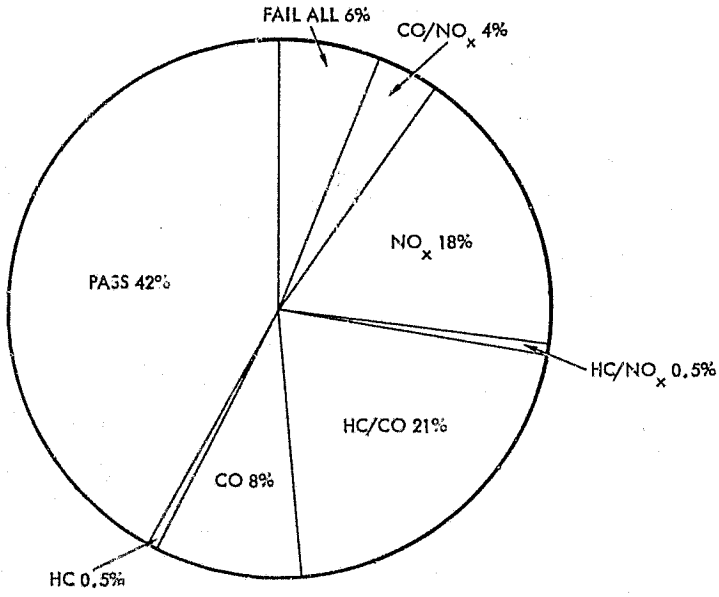


Fig. 2-9. Pass/fail Outcomes of the Initial Test on 1975 and 1976 Vehicles (Average Mileage 8000 mi) (Ref. 2-4)

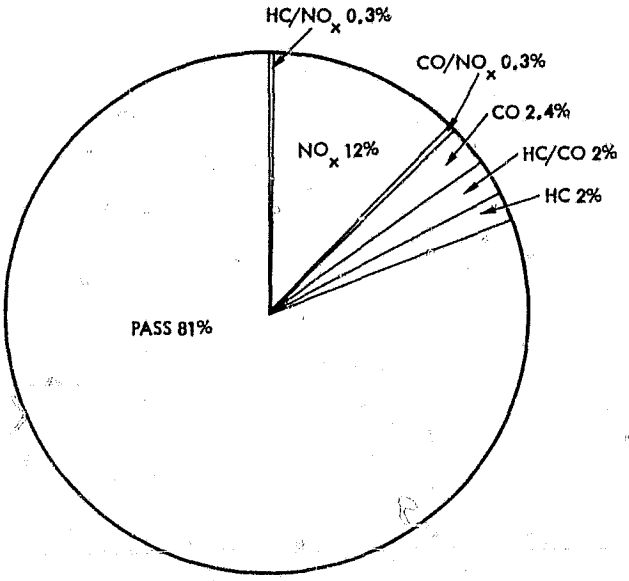


Fig. 2-10. Pass/Fail Outcomes of 1975 and 1976 Vehicles After the Restorative Maintenance Procedure was Complete (Average mileage 8000 mi) (Ref. 2-4)

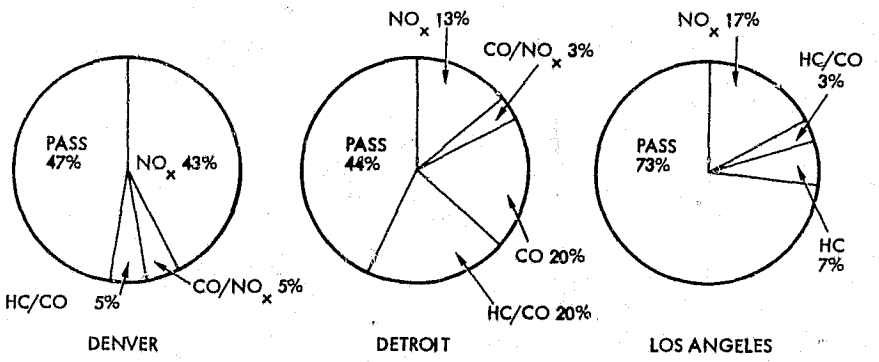


Fig. 2-11. Pass/Fail Outcomes of the Initial Test on 1977 Model Year Vehicles (<4000 mi vehicles) (Ref. 2-4)

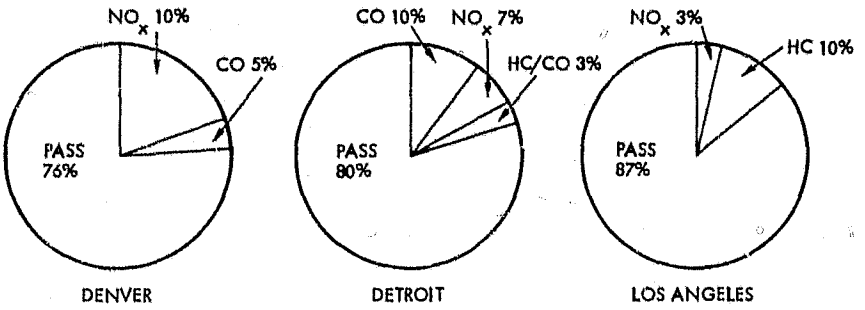


Fig. 2-12. Pass/Fail Outcomes of 1977 Vehicles After the Restorative Maintenance Procedure was Complete (<4000 mi vehicles) (Ref. 2-4)

The data presented on auto exhaust emissions clearly show that the average emissions are decreasing substantially with model year. The introduction of an I/M program has the effects of both decreasing average emissions and also increasing the uniformity of the emissions for a given model year. Thus this section has discussed the data base for determining what the emissions for LDV's are. An equally important consideration is to determine both the location and time at which the emissions take place. These estimates are made through the use of transportation network models, which forecast travel demand as a function of estimated urban and suburban activity. A discussion of the techniques used in these transportation models is presented in part B.4 of Appendix B.

By combining the exhaust gas emissions information by model year, type of vehicle, etc., with the estimates of the location of the vehicles as a function of time of day, estimates of the spatial and temporal distribution of emissions from mobile sources can be made. Of course these emissions must be added to the emissions from stationary sources to obtain estimates of the complete spatial and temporal distribution of emissions. Stationary source emissions are discussed in the following section in order to provide information on the current status of estimating the emissions from stationary sources.

Table 2-1. Percent of Emission Component Malperformance
(Ref. 2-4, p. 6)

Emission System	1975/1976 Models	1977 Models			
	Overall	Overall	Denver	Detroit	Los Angeles
Induction	5%	1%	5%	0%	0%
Carb/choke/exhaust heat control	66	47	52	60	30
Ignition	27	12	29	3	10
ECR -	15	6	14	0	7
Air pump	2	0	0	0	0
PCV	1	0	0	0	0
Exhaust	0	0	0	0	0
Evaporative emission control	1	0	0	0	0
Engine assembly/ miscellaneous	1	0	0	0	0
Any system	74%	58%	67%	60%	43%

Table 2-2. Percentage Emission Increases Due to Various Maladjustments and Disablements on 1975 and 1976 Vehicles (Ref. 2-4)

Type of Malperformance	No. of Cars	HC	CO	NO_x
Timing advanced 5°	36	24%	6%	19%
Idle mixture enriched	21	85	211	- 4
Manifold vacuum to distributor	14	36	29	11
Choke set 3 notches rich	22	23	80	15
ECR line plugged	37	21	71	123
Choke heater disconnected	12	30	127	- 7
Air pump deactivated	8	118	357	- 9
Selective malperformance*	30	86	230	175

*This term denotes those tests in which several of the above malperformances were present.

REFERENCES

2-1 Black, F., "The Impact of Emissions Control Technology on Passenger Car HC Emission Rate and Patterns," International Conference on Photochemical Oxidant Pollution and Its Control, Vol. 2, EPA 600/3-77-0016, Ecological Research Series, Research Triangle Park, N.C., 1053-1067, Jan. 1977.

2-2 Cantwell, Jr., E.N., et al., "Projections of Motor Vehicle Fuel Demand and Emissions," SAE 780933, E. I. Dupont De Nemours and Co., Inc., 1979.

2-3 Crowe, D., "Trends of California Urban Air Quality in the 1970's," California Air Resources Board Technical Services Division Aerometric Analysis Branch, Analysis and Research Section, Presented at the Conference on Air Quality Trends in the South Coast Air Basin, CIT, Pasadena, CA 21-22 Feb. 1980.

2-4 White, III, J. T., "An Evaluation of Restorative Maintenance on Exhaust Emissions from In-Use Automobile," SAE 780082, U.S./EPA, presented at Congress and Exposition, Cobo Hall, Detroit, Feb. 27 - Mar. 3, 1978.

2-5 Moore, L., and Unger, Jr., D.C., "A Study of Exhaust Emissions from 1975-1979 Model Year Passenger Cars in Los Angeles," Prepared for EPA, Emission Control Technology Division, EPA-460/3-79-004, Automotive Environmental Systems, Inc., 7300 Bolsa Ave., Westminster, CA 92683, July 1979.

SECTION 3

STATIONARY SOURCES

This section discusses the emissions from stationary sources, area sources, and point sources of HC, CO, SO_2, NO_x and TSP. It identifies the need for practicable test procedures and measurement methodology. In particular, this section examines the dominating stationary sources and compares their emissions with those from mobile sources for the South Coast Air Quality Management District (SCAQMD), Denver and Phoenix. The assessment of pollutant emissions from the stationary sources is needed for transportation planning in order to insure effective control strategies which satisfy ambient air pollutant standards. An effective assessment of pollutants from stationary sources must consider both the temporal and spatial distributions of the source emissions, and the uncertainties of these estimates.

3.1 WORK BREAKDOWN STRUCTURE

Stationary sources that significantly influence total pollutant emission inventory are outlined in Fig. 1-4. Both area and point sources are identified. The detailed listing outlines only the combustion point sources because of their dominance of stationary emissions. It should be noted that all significant sources of air pollutant emissions are considered in the development of an emissions data base for each air quality region. Each source is evaluated in order to determine its relative contribution. A comparison of the total emissions of particulates, SO_2, NO_x, HC and CO for the United States is presented in Tables 3-1 and 3-2 (Ref. 3-1). The comparison in Table 3-2 indicates substantial emissions of HC and NO_x come from both stationary and mobile sources. Fig. 1-2 shows the percent of total emissions from mobile sources for the South Coast Air Quality Management District (SCAQMD), Denver and Phoenix. From this data, it is seen that each of these regions has its own stationary source pollutant concerns. For example, in the SCAQMD, about 85% of the SO_x and 55% of the TSP is contributed by stationary sources.

Table 3-1. Estimated Pollutant Emissions in the United States, 1970 Through 1977 (millions of metric tons)

Year	Suspended Particles	Sulfur Oxides	Nitrogen Oxides	Hydro-Carbons	Carbon Monoxide	Total
1970	22.2	29.8	19.6	29.5	102.2	203.3
1972	19.6	29.6	21.6	29.6	103.8	204.2
1974	17.0	28.4	21.7	28.6	99.7	195.4
1976	13.2	27.2	22.8	28.7	102.9	193.8
1977	12.4	27.4	23.1	28.3	102.7	193.9

*Volatile hydrocarbons only; methane and other nonreactive compounds omitted as far as possible.

Table 3-2. Estimated Pollutant Emissions by Source, 1977 (millions of metric tons)

Source	Suspended Particles	Sulfur Oxides	Nitrogen Oxides	Volatile Hydro-Carbons	Carbon Monoxide
Transportation (autos, trucks)	1.1 9%	0.8 3%	9.2 40%	11.5 41%	85.7 83%
Combustion (power heating)	4.8 39%	22.4 82%	13.0 56%	1.5 5%	1.2 1%
Industrial Processes	5.4 43%	4.2 15%	0.7 4%	10.1 36%	8.3 8%
Solid waste (incinerators)	0.4 3%		0.1	0.7 2%	2.6 3%
Miscellaneous (fires, solvents	0.7 6%		0.1	4.5 16%	4.9 5%
Total	12.4	27.4	23.1	28.3	102.7

The methodology used and the uncertainties in this data using SCAQMD, Phoenix and Denver as examples are reviewed by examining the uncertainties in (1) the sources, (2) collected inventories, (3) models, and (4) predictions. An examination is presented of how these uncertainties affect (1) the mobile sources emissions inventories and modeling needed to predict ambient air quality and (2) the regulations of the pollutants from stationary and mobile sources. Furthermore, an appraisal of the existing data on stationary sources can be used to determine how uncertain the data might be and yet be effective for analyzing a given air quality region. In order to develop and verify air quality models, certain regions may require considerably more data than others. For example, the overall impact of all pollutant sources in the SCAQMD requires a considerably larger data base than that appropriate for Phoenix. In the case of Phoenix, a simple model analysis and knowledge of stationary point sources and their small contributions to CO and HC is adequate for developing requirements and demonstrating compliance with regulations.

3.1.1 Point Sources

The major stationary point sources are fossil fuel utilities and industrial plants. These sources generally have specific operating patterns and resultant emission histories. Knowing these patterns, daily or hourly emissions can be estimated on the basis of annual emissions. The yearly or hourly emissions from these sources are typically either based on measured actual emissions used to derive the needed inputs or estimated by emission modeling techniques. Emissions data derived from these approaches, e.g., using long-term average emission estimates for determining hourly emissions from stationary sources, are frequently highly uncertain.

3.1.2 Area Sources

The major stationary area sources that produce pollutants through combustion include residences, commercial, institutional and small manufacturing establishments. These sources burn fuels primarily for source heating, steam generating, cooking, and burning wastes. Again these sources exhibit patterns that vary diurnally, seasonally and with the ambient environment. Peaks of fuel consumption occur over a 1-3 hour period with different peaks from different sources. Knowing these patterns is essential to accurately estimating emissions from each of the identified areas.

The degree of knowledge of the emission sources determines the accuracy of the estimates. One current way of developing an emissions inventory for stationary sources is to apply emission factors which are characteristics of the combustion equipment. The emissions are determined by multiplying the emission factor by the quantities of fuel consumed. The accuracy of this type of estimate may be limited by a lack of knowledge of the concentration of the pollutants in the fuel.

3.2 STATIONARY SOURCES EMISSION FACTORS

Stationary sources emission factors have been compiled by EPA and are tabulated in the current revision of AP-42 and supplements (Ref. 3-2). This compilation of air pollutant emission factors contains emission data obtained from source tests, material balance studies and engineering estimates. The factors are continuously under revision to improve the emissions factor data and its accuracy. Extensive compilations of factors for stationary sources are included. The lack of agreement among the investigators regarding data formatting, the shortcomings in measurements, and incomplete sets of measured data requires various assumptions to be made in order to compile the factors (Ref. 3-2). The emission factors are current best estimates which qualitatively identify the uncertainties in the data set. However, they often do not provide for establishing upper or lower limits on the estimates. The uncertainty errors are expected to be important when predictions of ambient air quality in various regions are used to plan strategies. While the uncertainty may not affect establishing the qualification of a single stationary source under average emission conditions in order to meet the Federal and/or State standards, it will influence ambient air quality if the claim is that all stationary sources met the standards based on average emission factors.

Practicable test procedures are needed to demonstrate that stationary sources will meet the regulations based on both the source and air quality standards. Test procedures must be established and demonstrated to produce sufficiently accurate measurement of stationary sources. The measurements serve as an input to the models for predicting ambient air quality.

The uncertainties in the methods for measuring the constituents in the ambient air are discussed in Section 5. The current measurements for SCAB (LA), Denver and Phoenix indicate numerous violations of the standards for many of the pollutants (Refs. 3-3, 3-4 and 3-5). In particular, stationary sources account for most of the particulates violations in SCAB. For TPS the violations were so drastic in 1976 in SCAB that the uncertainty on the measurements was not important. However, for other contaminants, the data for SCAB in Table 3-3 shows that measurement uncertainty may be crucial in demonstrating that the standard is satisfied. In order to evaluate the air quality of the region, the emissions measurements and estimates must be correlated with the air quality measurements at the network stations in each region. This is being done in many of the areas, and uncertainties in the measured data and modeled results have been revealed. The importance of this correlation has been identified by EPA in the workshop on Requirements for Non-Attainment Area Plans. The basic areas identified for further work include need for more data and better models. Data are needed showing:

(1) Variations over time.
    Averaging time data samples.
    Variations by year compilations.

(2) Variations with monitor location.

Table 3-3. Most Stringent Air Quality Standards and Maximum Concentrations for 1976 (Ref. 3-3)

Contaminant	Most Stringent Standard	SCAB 1976 Maximum Reading
Photochemical oxidants (O_3)	0.08 ppm 1-hour average	0.38 ppm
Carbon monoxide (CO)	9.0 ppm 8-hour average	26.3 ppm
Nitrogen dioxide (NO_2)	0.25 ppm 1-Hour average	0.53 ppm
	0.05 ppm AAM	0.076 ppm
Sulfur dioxide (SO_2)	24-hour average	0.100 ppm
Particulate matter (TSP)	100 $\mu g/m^3$ 24-hour average	1988 (601)* $\mu g/m^3$ 1363 (424) $\mu g/m^3$
Sulfate (SO_4)	25 $\mu g/m^3$ 24-hour average	44.3 $\mu g/m^3$
Lead (Pb)	1.5 $\mu g/m^3$ monthly average	10.04 $\mu g/m^3$
Visibility	Number of days with visibility less than ten miles with relative humidity less than 70%	261 days

*Number in parentheses is the second highest value for the same station

On June 19, 1977, the ARB adopted a new state standard for sulfur dioxide of 0.05 ppm/24-hour average, occurring in combination with a violation of the state oxidant or TSP standards.

Model methodology should require:

(1) Measured data compatible with dispersion models where necessary.

(2) Data that accommodates temporal and meteorological anomalies.

(3) Consistency in emissions inventories and uncertainties.

3.3 STATIONARY SOURCES EMISSION INVENTORY

3.3.1 SCAQMD Emissions

Summaries of the current estimates of typical emissions from stationary sources for a summer day in 1975-76 are projected for a comparable day in 1982 and 1987 in Tables 3-4, 3-5 and 3-6. These summary tables provide comparisons of the relative percentage contribution and resultant importance of stationary sources vs other sources of emissions. For the SCAB, an effective stationary sources emissions inventory system is necessary for all pollutants except CO throughout the next 10 years because of their significant contributions. The reasonable further progress anticipated involves RHC reductions of 1/3 and NO_x reductions of 1/10 of the current amounts emitted.

The total reductions required must be examined critically to sort out the relative importance of stationary sources. Then the large stationary source contributors must be estimated, including the expected uncertainties in the emissions estimates measurements.

3.3.2 Denver System

Denver has had the same constraints as SCAQMD in predicting air quality levels, namely:

(1) Availability of meteorological, emissions and monitored air quality data.

(2) Development or access to models and modeling technologies necessary for the level of sophistication needed by Denver.

As described in Ref. 3-8 the analysis tools have been greatly facilitated by using the Systems Applications Inc. (SAI) Denver model for predicting CO and O_3. However, at the time of the study in late 1978 the reliability of models (SAI and Climatological Dispersion Model) to predict the reliability of NO_2 accurately has not been demonstrated quantitatively because the reactive properties of the NO_x is not accommodated.

Table 3-4. Base Year Emissions, 1975-76, by Major Source Category for an Average Summer Weekday in SCAB (Ref. 3-4)[a]

SOURCE	THC			RHC			CO		NO_x		SO_x		Particulate	
	TONS/ DAY	% of Man-Made	% of TOTAL	TONS/ DAY	% of Man-Made	% of TOTAL	TONS/ DAY	% of TOTAL	TONS/ DAY	% of TOTAL	TONS/ DAY	% of TOTAL	TONS/ DAY	% of TOTAL
STATIONARY (Area + Point)	676	38.9	23.5	510	34.5	30.1	215	2.6	464	36.2	313	81.9	150	56.2
On-Road Mobile	969	55.8	33.8	884	59.8	52.2	7699	91.2	694	54.1	37	9.7	94	35.2
Off-Road Mobile	92	5.3	3.2	84	5.7	5.0	527	6.2	125	9.7	32	8.4	23	8.6
Subtotal (Man-Made)	1737	100.0		1478	100.0		8441	100.0	1283	100.0	382	100.0	267	100.0
Natural Sources*	1132		39.5	215		12.7								
TOTAL	2869		100.0	1693		100.0	8441	100.0	1283	100.0	382	100.0	167	100.0

[a]Includes vegetation, landfills and animal waste.

Table 3-5. Projected Emissions, 1982, by Major Source Category for an Average Summer Weekday in SCAB (REf. 3-4)

SOURCE	THC			RHC			CO		NO_x		SO_x		Particulate	
	TONS/ DAY	% of Man-Made	% of TOTAL	TONS/ DAY	% of Man-Made	% of TOTAL	TONS/ DAY	% of TOTAL	TONS/ DAY	% of TOTAL	TONS/ DAY	% of TOTAL	TONS/ DAY	% of TOTAL
STATIONARY (Area + Point)	531	44.8	22.9	361	37.9	30.9	236	4.7	490	43.2	296	81.0	173	65.3
On-Road Mobile	544	45.9	23.5	490	51.5	42.0	4192	82.9	503	44.4	33	9.2	65	24.3
Off-Road Mobile	110	9.3	4.7	101	10.6	8.7	629	12.4	140	12.4	36	9.8	28	10.4
Subtotal (Man-Made)	1185	100.0		952	100.0		5057	100.0	1133	100.0	365	100.0	266	100.0
Natural Sources[b]	1132		48.9	215		18.4								
TOTAL	2317		100.0	1167		100.0	5057	100.0	1133	100.0	365	100.0	266	100.0

[a]Assumes currently mandated rules and regulations.
[b]Includes vegetation, landfills and animal waste.
[c]These projections assume the nonavailability of significant amounts of natural gas for use in power plants. A more generous assumption on natural gas availability was used in the District's Sulfur Dioxide/Sulfate Control Study; this would result in lower projections of sulfur dioxide emissions.

Table 3-6. Projected Emissions, 1987, by Major Source Category for an Average Summer Weekday in SCAB (Ref. 3-4)[a]

SOURCE	THC			RHC			CO		NO_x		SO_x		Particulate	
	TONS/ DAY	% of Man-Made	% of TOTAL	TONS/ DAY	% of Man-Made	% of TOTAL	TONS/ DAY	% of TOTAL	TONS/ DAY	% of TOTAL	TONS/ DAY	% of TOTAL	TONS/ DAY	% of TOTAL
STATIONARY (Area + Point)	531	51.0	24.4	360	44.2	34.9	252	6.1	543	46.8	316	79.0	184	64.6
On-Road Mobile	399	38.3	18.4	354	43.4	34.4	3176	77.5	474	40.8	46	11.5	70	24.6
Off-Road Mobile	111	10.7	5.1	101	12.4	9.8	674	16.4	144	12.4	38	9.5	31	10.8
Subtotal (Man-Made)	1041	100.0		815	100.0		4102	100.0	1161	100.0	400	100.0	285	100.0
Natural Sources[b]	1132		52.1	215		20.9								
TOTAL	2173		100.0	1030		100.0	4102	100.0	1161	100.0	400	100.0	285	100.0

[a]Assumes currently mandated rules and regulations.
[b]Includes vegetation, landfills and animal waste.
[c]These projections assume the nonavailability of significant amounts of natural gas for use in power plants. A more generous assumption on natural gas availability was used in the District's Sulfur Dioxide/Sulfate Control Study; this would result in lower projections of sulfur dioxide emissions.

In Denver, an emissions inventory system for each pollutant is developed for existing and projected time periods for all major source contributors. This emissions system provides emission modeling inputs for the current analyses. The total tonnages and relative contributions for the major source categories for each pollutant are shown in Tables 3-7 through 3-10.

Table 3-7 shows that over one-half of the particulate loading is attributed to dust and sand emissions from automobile travel on paved roads. The total automotive contribution varies from 73 to 84 percent of all sources over the projected period. Construction activity and point sources account for most of the remaining tonnage. For Denver, note that the total particulate tonnage is expected to increase so that by 1985 the emission inventory is 60 percent larger than the 1974 inventory.

The daily source contributions for hydrocarbons is shown in Table 3-8. Again the automobile is the most significant hydrocarbon source in Denver, contributing about 80% of the hydrocarbons. Point sources, cleaners, gasoline service stations, and airports also contributed. The total tonnage from automobile sources is scheduled to decrease by about one-half from 1980 to 1985. This is due to the presumed effectiveness of the Federal Motor Vehicle Control Program (FMVCP) (Ref. 3-5). Automotive sources contribute 44 to 50% of the total, depending on the year analyzed. The total tonnage of nitrogen dioxide is shown to increase with time to 1985, due to increased automobile emissions. The carbon monoxide emissions for 1974 are shown in Table 3-10. Future years were not projected due to technical difficulties in obtaining automotive contributions. The predominant contributor of CO emissions in 1974 is the automobile.

3.3.3 Phoenix Systems

The emissions inventory system is compiled annually by the Maricopa County Bureau of Air Pollution Control. The methodology used to calculate emissions from individual sources for the 1977 comprehensive emissions inventory is described in Ref. 3-7. Point sources dominate the stationary source emissions except for TSP. Unpaved roads and other miscellaneous area source contributions represent more than 80% of the total particulate emissions.

Mobile sources contribute 74% of the NO_x, 37% of the SO_x, less than 1% of the TSP (through exhaust), 95% of the CO and 56% of the NMHC. Tables 3-11 and 3-21 provide a more detailed breakdown of uncontrolled CO and NMHC emission inventories for Phoenix.

The reduction in CO and NMHC emissions between 1977 and 1985 in Tables 3-11 and 12 is due entirely to the Federal New Car Emission Control Program. It has been stated that ongoing inspection and maintenance and vapor recovery nonattainment area programs in Arizona are reducing emissions more significantly than indicated in these tables.

Table 3-7. Source Contributions in Denver, Particulate (Ref. 3-4)

	1974		1980		1985	
	Total tons/yr	% contribution	Total tons/yr	% contribution	Total tons/yr	% contribution
Space heat	511	1	631	1	678	1
Point projections	641	-	912	1	1,479	2
Automotive	4,213	7	4,477	6	5,622	6
Airports	63	0	131	0	76	0
Construction	8,730	15	8,730	12	6,978	8
Unpaved roads	6,010	10	6,010	8	6,010	6
Street sanding	15,857	27	22,690	31	32,634	35
Paved roads	16,543	29	23,656	33	34,046	37
Point sources	5,981	10	4,963	7	4,963	5
	57,908	99	72,200	99	92,486	100

Table 3-8. Source Contributions in Denver, Hydrocarbons (Ref. 3-4)

	1977		1980		1985	
	Total tons/yr	% contribution	Total tons/yr	% contribution	Total tons/yr	% contribution
Automobile, link	197.36	83.1	204.37	85.3	119.20	76.5
Automobile, area	1.82	0.8	0.78	0.3	0.76	0.5
Space heat	0.48	0.2	0.54	0.2	0.59	0.4
Oil paint	2.08	0.9	0.21	0.1	-0-	-0-
Gas stations	11.67	4.9	7.62	3.2	7.67	4.9
Cleaners	4.94	2.1	5.32	2.2	5.82	3.7
Incinerators	0.11	0.0	0.11	0.0	0.11	0.1
Point sources	13.14	5.5	13.97	5.8	14.55	9.3
Airports	5.92	2.5	6.61	2.7	7.27	4.7
	237.52	100.0	239.53	99.8	155.97	100.0

Table 3-9. Source Contributions in Denver, NO_2 (Ref. 3-4)

	1974		1980		1985	
	Total tons/yr	% contribution	Total tons/yr	% contribution	Total tons/yr	% contribution
Space heat	5,348	7.4	6,449	7.5	6,900	7.1
Small points	87	0.0	3,449	4.0	5,782	6.0
Automotive	31,646	43.7	40,290	47.0	48,167	49.8
Airports	1,552	2.1	1,836	2.1	2,104	2.2
Point sources	33,773	46.6	33,773	39.4	33,773	35.0
	72,406	99.8	85,797	100.0	96,826	100.0

Table 3-10. Source Contributions in Denver, CO, 1974 (Ref. 1-4)

Source Category	Total Tons/Day	% Contributions
Automotive	2925.0	94.5
Space heat	2.9	0.1
Point sources	167.6	5.4
	3095.5	100.0

In this section the emissions inventories for the South Coast Air Basin, Denver and Phoenix have been summarized. We were not able to obtain estimates of the precision of these emission estimates. These air quality management districts felt that it would be useful to have such estimates. However, they were not in a position to make estimates of the uncertainties in their emissions inventory data. Hence estimates of uncertainties in the emissions inventories are themselves uncertain.

Now that we have discussed the emissions inventories for these regions, it is appropriate to examine the air quality models that were used by the regions. This examination is performed in the next section.

Table 3-11. CO Emission Inventories Used in NAP Analysis (Ref. 3-7)[a]

	CO Emissions, tons/winter day				
	1975	1977	1980	1982	1985
Traffic emissions					
Light duty gas autos	756.78	565.65	506.41	407.51	296.57
Light duty gas trucks	229.25	196.61	172.36	148.67	142.80
Heavy duty gas vehicles	128.42	122.47	115.53	106.45	108.48
Diesel vehicles	3.75	4.10	4.11	4.36	4.67
Motorcycles	3.28	3.10	3.72	3.72	3.67
Airports	34.82	35.28	35.96	36.50	37.30
Railroads	3.04	3.11	3.22	3.31	3.44
Large point sources	0.62	0.61	0.60	0.59	0.58
Area sources					
Residential	0.98	1.04	1.14	1.20	1.28
Commercial/institutional	1.23	1.32	1.47	1.56	1.70
Industrial	4.78	5.18	5.76	5.78	5.80
Miscellaneous	0.99	1.05	1.13	1.20	1.30
Total emissions	1,167.94	939.52	851.41	720.85	607.59

[a]Future estimates assume no credit for NAP control strategies.

Table 3-12. NMHC Emission Inventories Used in NAP Analysis (Ref. 3-6)[a]

	NMHC Emissions, tons/summer day				
	1975	1977	1980	1982	1985
Traffic emissions					
Light duty gas autos	84.34	75.46	66.07	53.73	38.50
Light duty gas trucks	32.52	30.54	25.43	20.13	16.27
Heavy duty gas vehicles	18.09	16.86	13.69	10.43	9.97
Diesel vehicles	0.70	0.77	0.80	0.84	0.92
Motorcycles	0.86	0.83	0.81	0.84	0.81
Airports	5.82	5.83	5.86	5.88	5.91
Railroads	2.16	2.21	2.29	2.35	2.44
Large point sources	18.49	18.48	18.47	18.47	18.46
Area sources					
Commercial/institutional	0.23	0.24	0.27	0.29	0.32
Industrial	1.40	1.47	1.68	1.68	1.70
Miscellaneous					
Gasoline handling	32.99	34.15	37.61	39.85	43.22
Solvent evaporation	33.70	32.48	28.81	26.11	22.07
Pesticides, heaters and burning	3.53	3.45	3.21	3.09	2.91
Total Emissions	234.83	222.77	205.00	183.69	163.50

[a]Future estimates assume no credit for NAP control strategies.

REFERENCES

3-1 National Air Quality Monitoring and Emissions Trend Report, 1977, Environmental Protection Agency, December 1978.

3-2 Compilation of Air Pollutant Emission Factors, Third Edition (Including Supplements 1-7) AP-42 Part A, US/EPA, Office of Air and Waste Management, Office of Air Quality Planning and Standards, Research Triangle Park, N.C. 2711, August 1977.

3-3 "Draft Air Quality Management Plan," SCAG, Los Angeles, CA 90005, and SCAQMD, El Monte, CA 91731, October 1978 and Appendices, January 1979.

3-4 Griffin, R., "SIP Revision, Model Analysis of Existing and Future Air Quality Levels in the Denver Metro Areas," Air Pollution Control Division, Colorado Department of Health.

3-5 "Nonattainment Area Plan for Total Suspended Particulates," Maricopa County Urban Planning Area, AC:AAA-295, Prepared by Technical Staff Bureau of Air Quality Control, Division of Environmental Health Services, Arizona Department of Health Services, June 1979 (Revised October 15, 1979).

3-6 Littman, F.E., et al., "A Regional Air Pollution Study (RAPS) Preliminary Emission Inventory," prepared for US/EPA Office of Air and Water Programs, Air Data Branch, Durham, NC, SRI Project EGU-2579, Stanford Research Institute, Menlo Park, CA, January 1974.

3-7 "Nonattainment Area Plan for Carbon Monoxide and Photochemical Oxidants," Maricopa County Urban Planning Area, AC:AAA-300, prepared by Technical Staff Bureau of Air Quality Control, Division of Environmental Health Services, Arizona Department of Healt Services, December 1978 (Rev. February 16, 1979).

3-8 "Analysis of Air Quality in the Denver Air Quality Maintenance Area," The Air Pollution Control Division, Department of Health, Colorado, March 1977.

SECTION 4

AIR QUALITY MODELING

Air quality models are used to relate the emissions of air pollutants, from a variety of sources, to the ground level concentrations of each pollutant. These models require input information on meteorology and emissions inventories. Some math models may require the use of a set of chemical equations to describe the chemistry taking place in the atmosphere. These inputs to an air quality simulation model have been schematically represented in Fig. 1-3. In this section, these inputs, together with the corresponding math model, are discussed. The characteristics of the following types of air quality simulation model are examined:

(1) Linear rollback

(2) Empirical Kinetic Modeling Approach (EKMA).

(3) Photochemical dispersion models.

4.1 METEOROLOGY

According to Slade (Ref. 4-1):

> The waste products of our civilization must be disposed of. Receptacles for this debris are the earth's land masses, water bodies, and the atmosphere. The science of meteorology is important in the study of the disposition of waste products in the atmosphere.
>
> Wastes that are released to the atmosphere consist of particles and gases. Atmospheric residence times for some of these materials may be very short, hours or even minutes. For other materials residence times may be measured in terms of years. Regardless of the residence time, the movement of gases and particles in the atmosphere will be, in large measure, governed by the motions of the atmosphere. Some atmospheric motions dictate the paths to be followed by airborne contamination; other motions determine the extent to which the contaminants will be diluted. The study of the effects of atmospheric motions on suspended pollutants is a branch of the science of meteorology variously categorized under the headings "air-pollution meteorology" or "atmospheric diffusion."

Meteorological factors have a very large impact on the relationship between air pollutant emissions and resultant air quality. For example, in the South Coast region the ozone concentrations may change from a value which meets the national ambient air quality standard (.12 ppm hourly average) to a value several times higher from one day to the next. These dramatic changes in air quality may take place with emissions which are essentially the same. Some meteorological factors which have a significant influence on air quality predictions include:

(1) Three-dimensional distribution of wind speed and direction.

(2) The topography of the land.

(3) Pollutants remaining from the previous day.

(4) Background pollutant effects.

(5) Cloud cover.

(6) Temperature distribution.

These factors are discussed in the following text.

4.1.1 Wind and Temperature Measurements

Wind measurements often determine only the horizontal wind direction and wind speed at ground level. In many areas the only wind measurements may be from the local airport. Some areas such as the South Coast Air Basin are characterized by many such wind stations, e.g., 50 stations. In areas where special studies are being performed wind measurements may include the verticle component of both wind speed and direction. Such measurements were performed in EPA's Regional Air Pollution Study (RAPS).

Measurements of wind speed and direction aloft are routinely performed at airports twice daily through the release of pilot balloons. More frequent measurements can be made during special studies. In some cases the pilot balloons are equipped with a temperature sensor and a transmitter. In these cases, measurements of temperature as a function of altitude are obtained. In special cases such as the Regional Air Pollution Study the above temperature and wind measurements may be supplemented by additional measurements from fixed wing aircraft and helicopters.

The wind speed and direction are often recorded on charts. Hourly averages of these speeds and directions are kept as part of the permanent record. Hence if the wind speed and direction were highly variable during a given hour, the record would only indicate the average condition. Any air quality simulation model that used this averaged data would have already filtered out considerable variability which could have a highly significant impact on the error in estimating air pollutant concentrations. This class of error is especially significant in estimating hourly averages of pollutant concentrations.

It should be pointed out that this type of error is intrinsic to all the air quality simulation models. The photochemical dispersion models may estimate wind speed and direction on a 5-minute basis from the hourly averaged data. However, the accuracy of these wind representations cannot be known unless examination is made of the detailed charts rather than the hourly averages. Furthermore, it would be necessary to check the model prediction with both the 5-minute wind values and the hourly averages to determine the size of the error associated with this type of averaging. These types of test are being examined in the Regional Air Pollution Study (RAPS) analysis. We have not been able to obtain the results of such analysis in this study.

There may be inherent physical limitations associated with modeling the influence of meteorological factors at time scales of the order of 5 minutes or less. For example the Pasquill stability categories (Ref. 4-2) that are used for both gaussian plume modeling and other forms of modeling are intended to represent average dispersion over a 15-minute interval if the direction of the wind is fairly constant. If the wind changes directions by, for example, 90° within the 15 minutes, these representations are of doubtful use.

Similar comments can be made about the influence of the frequency of vertical temperature distribution measurements (Ref. 4-3).

4.1.2 Complex Terrain

The topography of the land is quite complex for both the Denver Region and the South Coast Region. In comparison, Phoenix is relatively simple. For example, a description of the topography for the South Coast Air Basin (SCAB) follows (Ref. 4-3):

> The Basin is a coastal plain with connecting broad valleys and low hills bounded by the Pacific Ocean to the southwest and high mountains along the perimeter (see Fig. 4-1).
>
> Inland, the SCAB terrain varies from rugged, high mountains with elevations up to 11,500 feet, mainly in the northern and eastern parts of the area, to almost flat-lying coastal plans at or just above sea level. Between the mountains and the sea is a complex array of terraces, hills, and foothills.

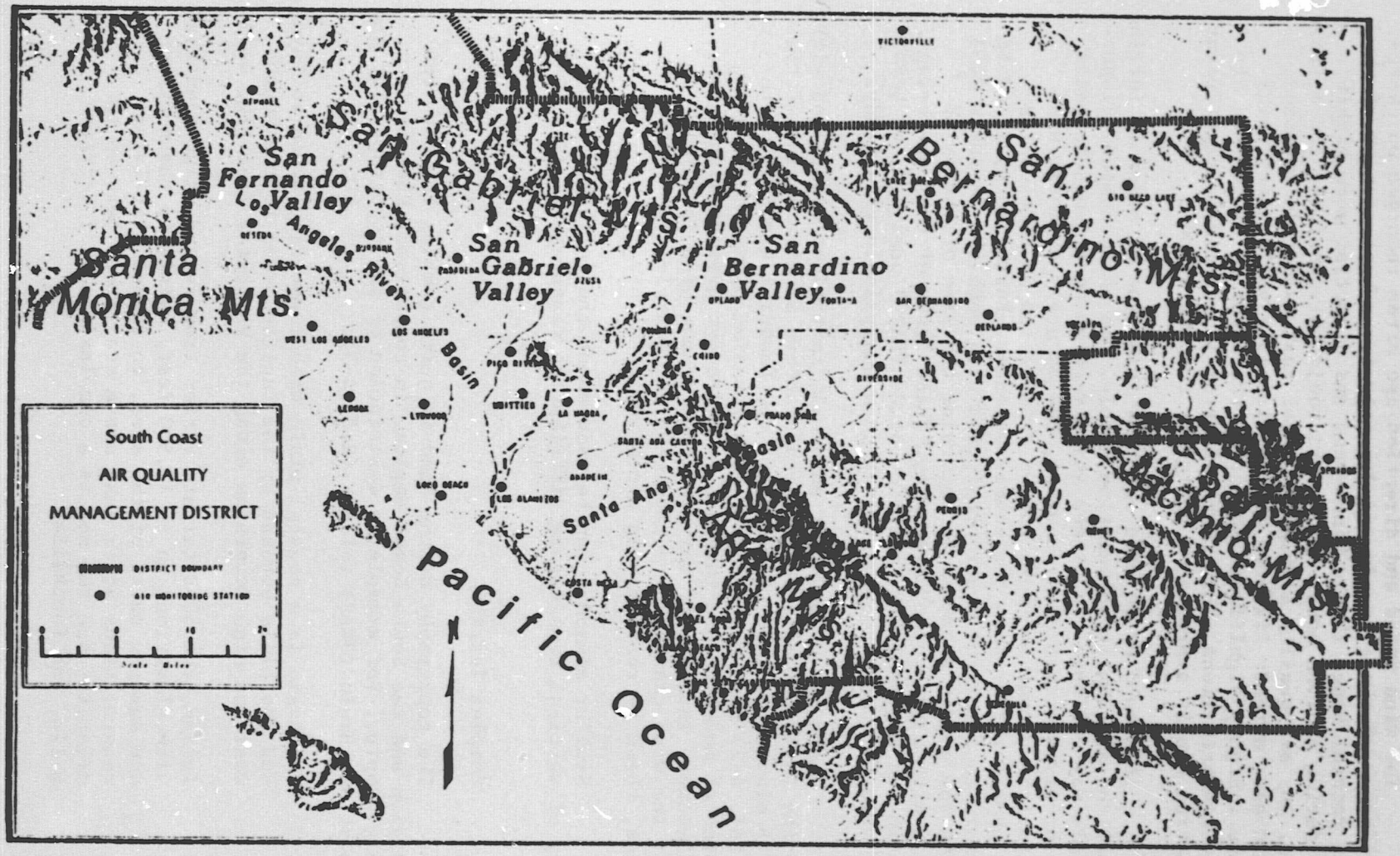


Fig. 4-1. Topographic Features / SCAB

The SCAB area includes portions of two geomorphic provinces, the Transverse Ranges (Santa Monica, San Gabriel and San Bernardino Mountains) and the Peninsular Ranges. The Transverse Ranges include mountains and valleys having a dominant east-west trend. The Peninsular Ranges have a northwest trend. Besides the two ranges, the Basin is also crossed by the lower lying Coastal Ranges (Santa Ana, Verdugo, and portions of the Santa Monica Mountains). Between the Pacific Ocean and the Coastal Ranges lie the Coastal Plains, and between the Coastal Ranges and the Transverse and Peninsular Ranges lie the inland valleys.

According to Ref. 4-4 the overall average of inversions, with heights less than 2,500 ft above sea level, occur 22 days each month. Since this basin extends approximately 100 miles on a side, the horizontal dimension is about 200 times the vertical dimension. Thus pollutant concentrations are highly nonuniform within this effectively wide shallow basin (Ref. 4-5). The complex set of mountains, foothills, canyons, etc., further complicates the distribution of air pollutants.

In the more complex models for air quality simulation the complexity of topography is included as a set of inputs. For example, a computerized topo-map of the South Coast Region is shown in Fig. 4-2. (Ref. 4-6)

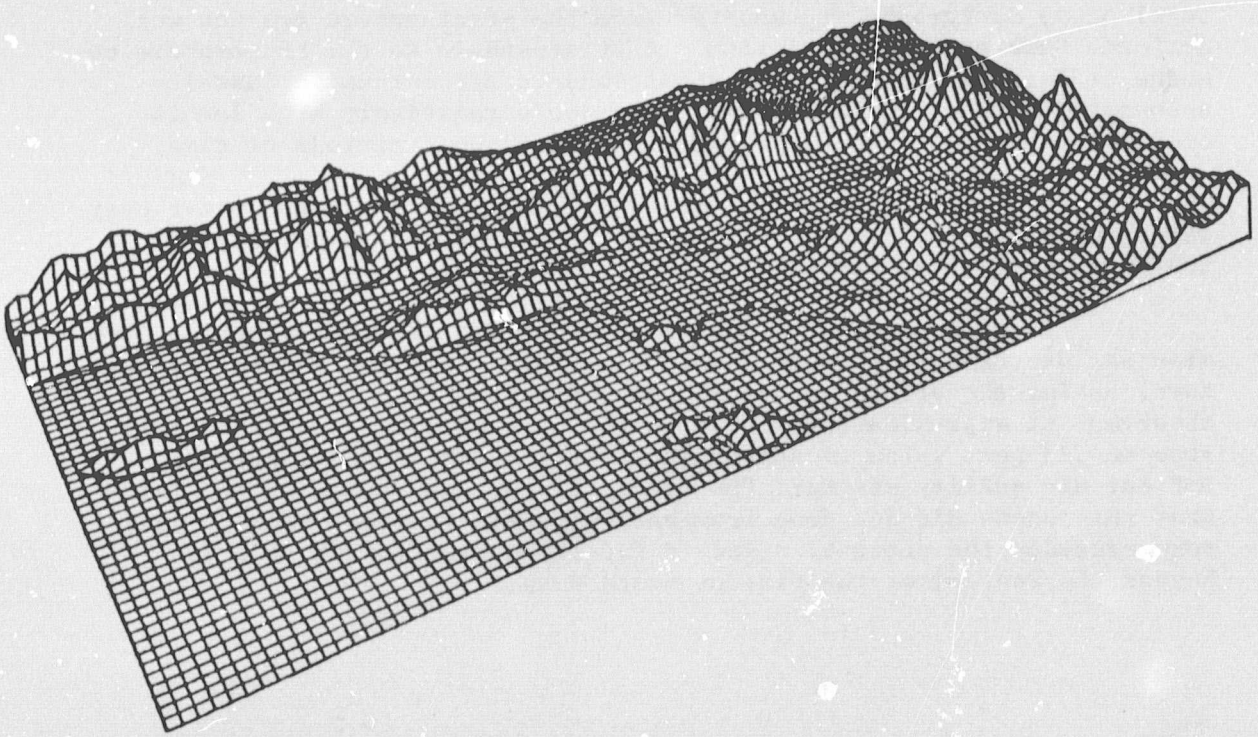


Fig. 4-2. Grid Structure Model of the South Coast Region

4.1.3 Background Effects

Background concentrations of pollutants are associated with two principal categories: (1) pollutants transported into a given region and (2) pollutants that are generated by natural earth processes such as windblown dust in the desert, or hydrocarbons emitted from trees. The second category is referred to as geogenic sources. Hence a given area might have certain concentrations of pollutants even if the area were totally uninhabited.

The pollutants transported into a given region can be a combination of natural background coming into the region plus pollutants from major sources such as metropolitan areas which are outside of a given region. The natural background pollutants may be assumed to be a constant for the purposes of worst-case analysis. The pollutants that are transported into a region from another urban area may decrease as the air quality in that other metropolitan area decreases. Hence, it is important to distinguish between transported pollutants from geogenic and anthropogenic sources.

Some geogenic sources may be approximately constant, whereas others may be highly variable. An example of a highly variable geogenic source is stratospheric intrusion. At the National Commission on Air Quality's Photochemical Modeling Workshop, it was reported that "Ozone concentrations in the troposphere free from anthropogenic influences are, according to some, generated primarily by ozone transfer to the troposphere from the stratosphere. The relative contributions of various stratospheric/tropospheric ozone transfer mechanisms to the total ozone background transported from the stratosphere are not well defined. However, transport from the stratosphere to the troposphere by means of large-scale eddies or stratospheric air intrusion, usually associated with cyclonic activity, can cause relatively high levels of ozone to be observed at ground levels for short periods of time."

"Ozone concentrations in excess of 80 ppb often occur over large regions where plumes from large urban centers are not apparent (rural ozone blanket problem)."

At the workshop, an extreme example of stratospheric intrusion was described for Santa Rosa, California, which is a rural farming town. During the afternoon, ozone concentrations of .04 ppm were observed. At approximately 4:00 a.m. the next morning, the ozone level rose to .23 ppm, which is approximately twice the national primary ambient air quality standard for ozone. It was apparently quite clear that this ozone did not come from anthropogenic sources. Hence, Santa Rosa exceeded the national standard for ozone even though this was beyond the control of the City of Santa Rosa.

It is apparent that methods are needed to allow for phenomena such as stratospheric intrusions to be considered in such a way as not to constitute a violation of the national primary air quality standard. One approach to this problem would be to determine the frequency on an annual basis of stratospheric intrusion of ozone, and to allow the standard to be exceeded by the number of days that such intrusion could occur. It should be noted that this argument has nothing to do with the health effects of ozone. It simply may be that even though ozone has health effects at some low concentrations, we may not be able to prevent those concentrations from occurring a few times a year.

This type of problem is not a major consideration in some places such as the South Coast Air Basin, where the ambient air quality standard for ozone is exceeded on over 200 days a year. However this type of phenomenon could be a problem in a region which exceeded the ozone air quality standard a few times a year. In addition, if some more robust standard were used, such as "not to exceed .12 ppm more than 10 times a year," then many of the instrumentation and modeling problems would be simplified. For example, the statistical basis of using the EKMA model would be greatly improved. The general influence of background pollutant concentration on the EKMA model is discussed in Ref. 4-7.

4.1.4 Multiple Day Meteorology

In some areas, such as the South Coast Air Basin and the Denver region, the wind may change direction by 180° from day to night. Hence, the emissions and their by-products from one day may be blown back to the region to contribute to degrading the next day's air quality. In some sense, under these circumstances a region can be "downwind of itself." This component of pollutants transported into a region may strongly influence the "boundary conditions" for a given day. Hence, a "worst case day" may not only be influenced by the emissions of that day, but also be influenced by the emissions of the previous day.

This concept was thought to have considerable importance by the members of the Photochemical Modeling Panel at the National Commission on Air Quality's Workshop. The following comments were made in an issue paper.

At night, atmospheric inversion protects ozone aloft from destruction by NO scavengers while ground level ozone tends to be destroyed (although high ground level ozone levels have been measured at night). The following day, inversion breakup allows downward mixing of the ozone and any unreacted precursors aloft with the new day's local precursor emissions and any other pollutants transported from upwind. Some investigators believe that the downward mixed ozone can contribute substantially to the new day's urban ozone levels; others believe that only ozone generated in the mixed layer is important--that ozone aloft will be scavenged by NO upon downward mixing into urban areas. Such may not be the case in rural areas where NO concentrations are low. Some also believe that the second day irradiation of unreacted precursors aloft may contribute substantially to the new day's ozone levels. In any event, the downwind (local) ozone levels are not necessarily primarily determined by local emissions.

In separate conversations with both G. McRae of Caltech and A. Ranzierri of the California Air Resources Board, they stated that on days of ozone episode conditions the background concentrations of air pollutants at the beginning of the day can have a highly significant influence on the air quality later in the day. From the comments above concerning downward mixing of ozone and any unreacted precursors aloft, it appears that it is important to include the influence of multiple-day meteorology in grid air quality simulations for ozone.

4.2 PHOTOCHEMISTRY

In order to provide some indication of the complexity of the interactions between transportation systems and the air quality of a region, this section discusses photochemical smog reactions. This discussion will not be complete since its intention is only to outline the principles involved in photochemistry in the urban atmosphere. For a more detailed presentation, please see Air Pollution: Physical and Chemical Fundamentals by John H. Seinfeld. The brief outline presented here attempts to show how the oxides of nitrogen, hydrocarbons and sunlight interact to form photochemical smog.

Many urban centers are characterized by a brown pall of gas surrounding the city. This brown gas is nitrogen dioxide. It absorbs sunlight for wavelengths less than 4300 A. This absorption process can cause the rupture of one of the bonds in the NO_2 molecule thus changing the NO_2 to NO plus an oxygen atom in the triplet P state, $O\ (^3p)$.

$$NO_2 + h\nu \rightarrow NO + O(^3p) \qquad (4\text{-}1)$$

This triplet p oxygen atom can collide with other oxygen molecules in such a way to produce ozone as is shown in Eq. (4-2).

$$O(^3p) + O_2 + M \rightarrow O_3 + M \qquad (4\text{-}2)$$

In this equation, M symbolizes a third molecule which absorbs excessive vibrational energy. Once the ozone is formed, it can react with nitric oxide, if it is available in the atmosphere, to once again form NO_2 and oxygen as outlined in Eq. (4-3).

$$O_3 + NO \rightarrow NO_2 + O_2 \qquad (4\text{-}3)$$

Thus, the NO_2 molecule in Eq. (4-2), which is broken up by the light into NO and oxygen, has the NO component converted back to NO_2 through an interaction with ozone. Hence, the chemical cycle is complete. This cycle is referred to as the photolysis cycle. Some further insight into photochemistry can be obtained by looking at the reaction kinetics, i.e., the rates at which these compounds interact. From Eq. (4-1), it may be shown that the rate of change in the concentration of nitrogen dioxide, d/dt $[NO_2]$ is proportional to the concentration of NO_2 as shown in Eq. (4-4).

$$\frac{d}{dt}[NO_2] = k_1 [NO_2] \qquad (4\text{-}4)$$

where k_1 depends on the intensity and wavelength of the sunlight.

From Eq. (4-3) an expression can be written for the rate of change in the concentration of ozone. Since the ozone and the NO interact in this equation, the rate of change of ozone is proportional to the product of the concentration of ozone and the concentration of NO or nitric oxide as is shown in Eq. (4-5).

$$\frac{d}{dt}[O_3] = k_3 [O_3] \quad [NO] \qquad (4\text{-}5)$$

When Eqs. (4-1,2,3) are in equilibrium, every molecule of NO_2 that is broken up is replaced by the breakup of a molecule of ozone. Hence, in equilibrium the time rate of change of the concentration of NO_2 is equal to the time rate of change of the ozone concentration as shown in the following.

$$\frac{d}{dt}[NO_2] = \frac{d}{dt}[O_3]$$

by substitution from (4) and (5), this becomes:

$$k_1 [NO_2] = k_3 [O_3] \quad [NO]$$

Hence, the concentration of ozone is shown in Eq. (4-6).

$$[O_3] = \frac{k_1}{k_3} \frac{[NO_2]}{[NO]} \qquad (4\text{-}6)$$

Thus, if Eqs. (4-1,2,3) govern completely the production ozone, the concentration of ozone depends on the ratio of concentration of NO_2 to the concentration of NO. According to Seinfeld, the cycle time associated with reactions 1 through 3 is typically short enough that this approximation for the steady-state concentration of ozone is reasonable.

Thus, to convert NO to NO_2 requires the consumption of a molecule of ozone. We will next examine chemical reactions which convert NO to NO_2 without the consumption of ozone. These reactions cause the ratio of NO_2: NO to become larger, hence the ozone concentration increases.

According to Falls et al. (Ref. 4-8), the two main processes by which NO is converted to NO_2, without the loss of ozone, involve the hydroperoxy radical NO_2 and peroxyalkyl radicals RO_2 via

$$HO_2 + NO \rightarrow OH + NO_2$$

$$RO_2 + NO \rightarrow RO + NO_2$$

Hydroperoxy and peroxyalkyl radicals arise in the photochemical smog system from the photolysis and oxidation of hydrocarbon species (Ref. 4-8). The contribution of non-methane hydrocarbon to photochemistry is essential in order to explain the observed nitric oxide to nitrogen dioxide conversions. In this cycle, NO is converted to NO_2 by interacting with hydrocarbons without reacting with ozone. Hence the concentration of NO_2 is increased without a corresponding decrease in the concentration of ozone. At the start of this cycle, hydroxyl radicals (OH·) and hydrocarbons (HC) react according to Eq. (4-7):

$$OH\cdot + HC \rightarrow R\cdot \qquad (4\text{-}7)$$

The radical R· reacts with an oxygen molecule to form a peroxy radical, RO_2:

$$R\cdot + O_2 + M \rightarrow RO_2\cdot + M \qquad (4\text{-}8)$$

Typically, the RO_2· converts NO to NO_2 and forms an oxyl radical, RO :

$$RO_2\cdot + NO \rightarrow RO\cdot + NO_2 \qquad (4\text{-}9)$$

The RO· reacts with oxygen to typically form a hydroperoxyl radical, HO_2· and a carbonyl compound, OHC:

$$RO\cdot + O_2 \rightarrow OHC + HO_2\cdot \qquad (4\text{-}10)$$

Finally, to complete the cycle, HO_2· reacts with NO to produce more NO_2 and to regenerate the OH· originally shown in Eq. (4-7).

$$HO_2\cdot + NO \rightarrow OH + NO_2 \qquad (4\text{-}11)$$

Hence, the influence of NO, NO_2 and HC on photochemical smog (O_3) can be understood in a simplified way through two sets of cyclic chemical reactions. The photolysis series (Eqs. 4-1 to 4-6) is typically fast enough for the steady state relationship (Eq. 4-6) to be valid. The second series (Eqs. 4-7 through 4-11) provides an example mechanism for conversion of NO to NO_2 without depleting O_3. According to Seinfeld, "Although this description is very simplified, these two cyclic series contain the essential features of ozone formation." These two types of reactions are shown in Fig. 4-3.

4.3 THREE AIR QUALITY SIMULATIONS MODELS

In the following section three air quality simulation models are discussed: linear rollback, EKMA, and photochemical disbursion. These models range in order of complexity from the simplest to the most complex in terms of both input data requirements and sophistication required to operate the models. Recent review of other models has been provided by Bruce Turner of the Environmental Protection Agency (Ref. 4-9). This review, coupled with the discussion paper that followed it (Ref. 4-10), provides real insights into the strengths and limitations of the models described in these reports.

4.3.1 Rollback

Linear rollback is by far the simplest air quality simulation model. It is based on the idea that the concentration of a given species of air pollutant in a given region is equal to the background concentration plus a term which increases linearly with the emissions of the pollutant species. This idea is expressed mathematically in Eqs. (4-12) and (4-13).

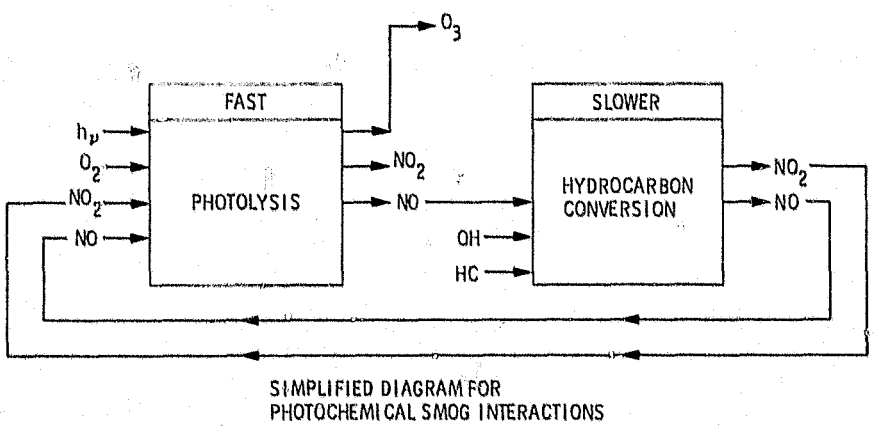


Fig. 4-3. Simplified Diagram for Photochemical Smog Interactions

$$\begin{matrix}\text{Air} \\ \text{Pollutant} \\ \text{Concentration}\end{matrix} = \begin{matrix}\text{Background} \\ \text{Air Pollutant} \\ \text{Concentration}\end{matrix} + \alpha \text{ Emissions, or} \qquad (4\text{-}12)$$

$$C_i = B_i + \alpha E_i \qquad (4\text{-}13)$$

If we know both the air quality and the emissions for the same period of time in the past, α may be eliminated from Eq. (4-13). By substituting this information in Eq. (4-13) we arrive at Eq. (4-14).

$$C_{i_{\text{Past}}} = B_i + \alpha E_{i_{\text{Past}}} \qquad (4\text{-}14)$$

Eq. (4-14) can be used to eliminate the proportionally constant α to arrive at Eq. (4-15), which predicts the air quality in the future in terms of the background, the emissions in the future, the emissions in the past and the air quality in the past.

$$C_{i_{\text{Future}}} = B_i + \left(C_{i_{\text{Past}}} - B_i\right)\left(\frac{E_{i_{\text{Future}}}}{E_{i_{\text{Past}}}}\right) \qquad (4\text{-}15)$$

The virtue of rollback is its extreme simplicity. This virtue is also its vice, since its simplicity causes many people to believe that it just simply is not adequate to handle air quality management. It is thought that this equation aggregates at too high a level. For example, the emissions in Eq. (4-15) represent the emissions for an entire air quality basin, and the air quality is some representative, perhaps average number, for the entire basin. These concepts ignore the idea that both the air quality and the emissions pattern can be highly dependent on position within the basin. Nevertheless, we shall examine the linear rollback approach as compared to the more sophisticated empirical kinetic modeling approach (EKMA) to explore the practical differences between the predictions of each model.

4.3.2 Empirical Kinetic Modeling Approach (EKMA)

The fundamental ideas behind the Empirical Kinetic Modeling Approach are described quite well in the users' manual (Ref. 4-11). The approach is based on the use of smog chamber data to relate ambient concentrations of non-methane hydrocarbons (NMHC), oxides of nitrogen (NO_x), and ozone. A standard set of EKMA curves is shown in Fig. 4-4. Depending on the sophistication that is used in the EKMA, either these standard curves can be used, or a set of special curves can be generated for a given location such as the South Coast Air Basin. Referring to Fig. 4-4, the vertical axis is the oxides of nitrogen, NO_x, in parts per million. The horizontal axis represents the non-methane hydrocarbons, NMHC, again in parts per million, as carbon. A set of curves

is drawn on these axes, each for a fixed value of ozone. For example, the curve which is marked .12 ppm of ozone indicates the focus of points of NO_x and NMHC which will produce a peak ozone hourly average value of .12 ppm, the National Primary Ambient Air Quality Standard for ozone.

As an example of the use of EKMA, suppose the concentration in a given locality is .28 ppm of ozone. Suppose in that locality it has been determined that the ratio of non-methane hydrocarbon to the oxides of nitrogen is 12 to 1. One can draw a line on the EKMA diagram with a slope of 1 part in 12. This line intersects the .28 curve line at the point marked A on the diagram. The point A corresponds to the following values: NMHC = 1.25 ppm and NO_x = .12 ppm.

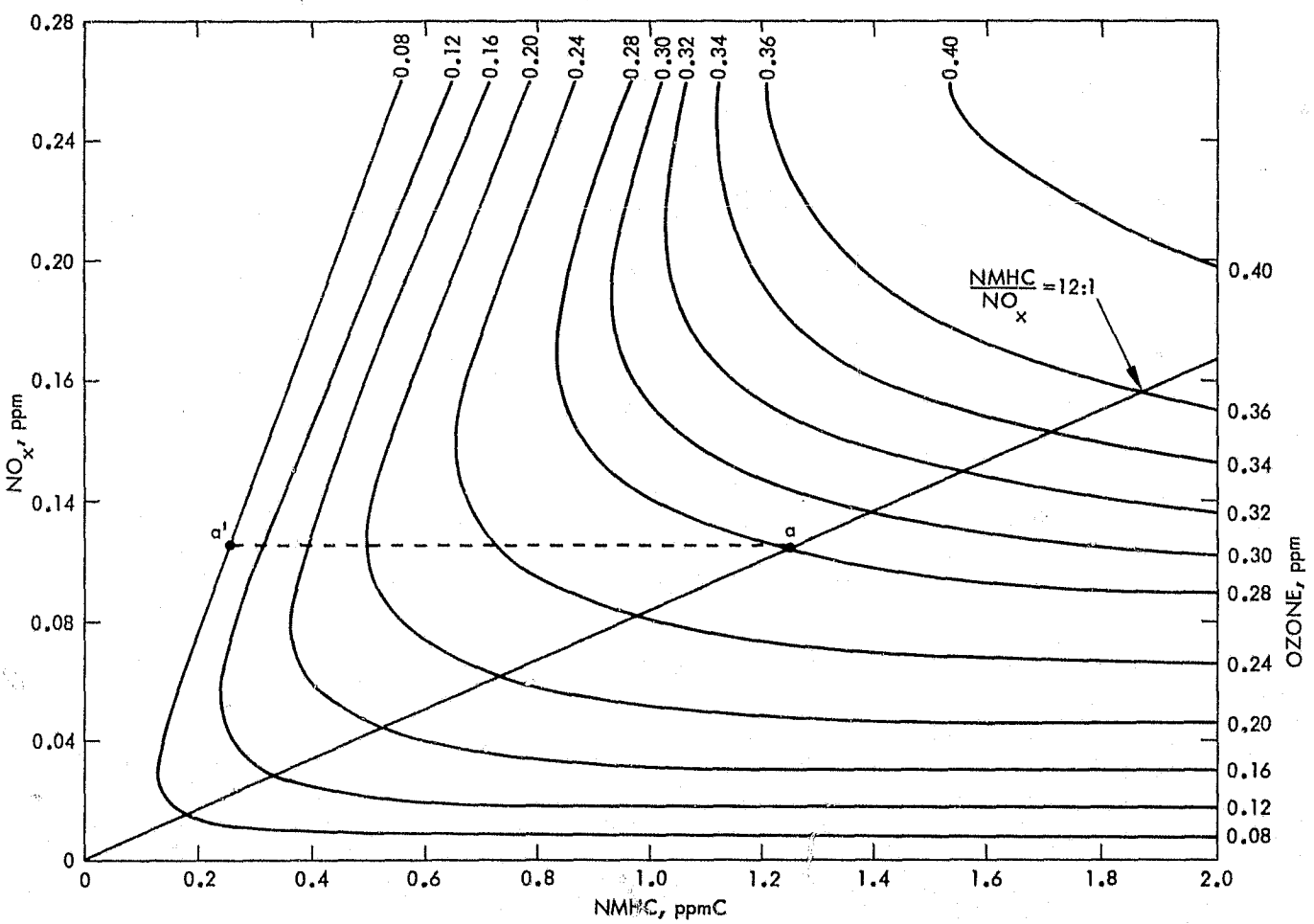


Fig. 4-4. Ozone Isopleths Derived for an EKMA Analysis

If one decides to reduce this peak value of ozone with a strategy which is based solely on reducing non-methane hydrocarbons, then one moves from the point marked A on the EKMA diagram to the point marked A'. Since the value of NO_x remains the same, the value of non-methane hydrocarbons is reduced to about 0.35 ppm; i.e., the non-methane hydrocarbons are reduced by approximately 72%. An opposite strategy could be followed to reach the ozone standard. The value of non-methane hydrocarbons is reduced by approximately 72%. An opposite strategy could be followed to reach the ozone standard. The value of non-methane hydrocarbons could be kept constant, and the oxides of nitrogen could be reduced. In this case, we move from point A to point B in Fig. 4-4. The value of NO_x is reduced from .12 ppm to a new value of 0.25 ppm; i.e., NO_x is reduced by 79%. Apparently other approaches can be used, such as simultaneously reducing both the NO_x and the NMHC. For example, one could move along the 12 to 1 ratio line shown in Fig. 4-4 until that line intersected the .12 ppm ozone isopleth.

Hence the EKMA approach can be used to estimate effects of the combined reduction in non-methane hydrocarbons and oxides of nitrogen. Since ozone depends on both NO_x and NMHC, as discussed in Section 4.2 on photochemistry, this approach appears to be intrinsically superior to linear rollback, which simply relies on reducing a single pollutant to reduce the ozone. To test this concept, we have compared the predicted percentage reductions in non-methane hydrocarbon using both linear rollback and the Empirical Kinetic Modeling Approach. For example, let's assume that both NMHC and NO_x are simultaneously reduced in such a way that their ratio NMHC/NO_x is a constant. The data for this comparison is provided in Ref. 4-12. Figure 4-5 shows the percentage reduction in NMHC determined by the EKMA approach on the vertical axis and the percentage reduction in NMHX as determined by linear rollback on the horizontal axis. If the two models completely agree, the data points would lie on the 45° line, which is marked "rollback equal EKMA." Data shown in Fig. D for ozone concentrations of .2, .3, and .4 ppm comes surprisingly close to the 45° line. This data could be represented by a line parallel to the 45° line with an offset of approximately 15%. That is, if we add 15% to the predicted reductions based on linear rollback, we arrive at a prediction which agrees quite closely with the EKMA reductions. Hence, in this mode, EKMA requires tighter controls on emissions than linear rollback. An alternate approach to reaching the ozone standard can be used to compare these models. In this approach, the NO_x is held constant and the NMHC is reduced. This is similar to that described before in Fig. 4-4, where the reduction went from point A to point B. A comparison of the reductions in non-methane hydrocarbon as determined by EKMA and linear rollback are shown in Fig. 4-6. Once again the 45° line represents the "line of equivalence" between the two models; i.e., on this line the two models predict the same percentage reductions.

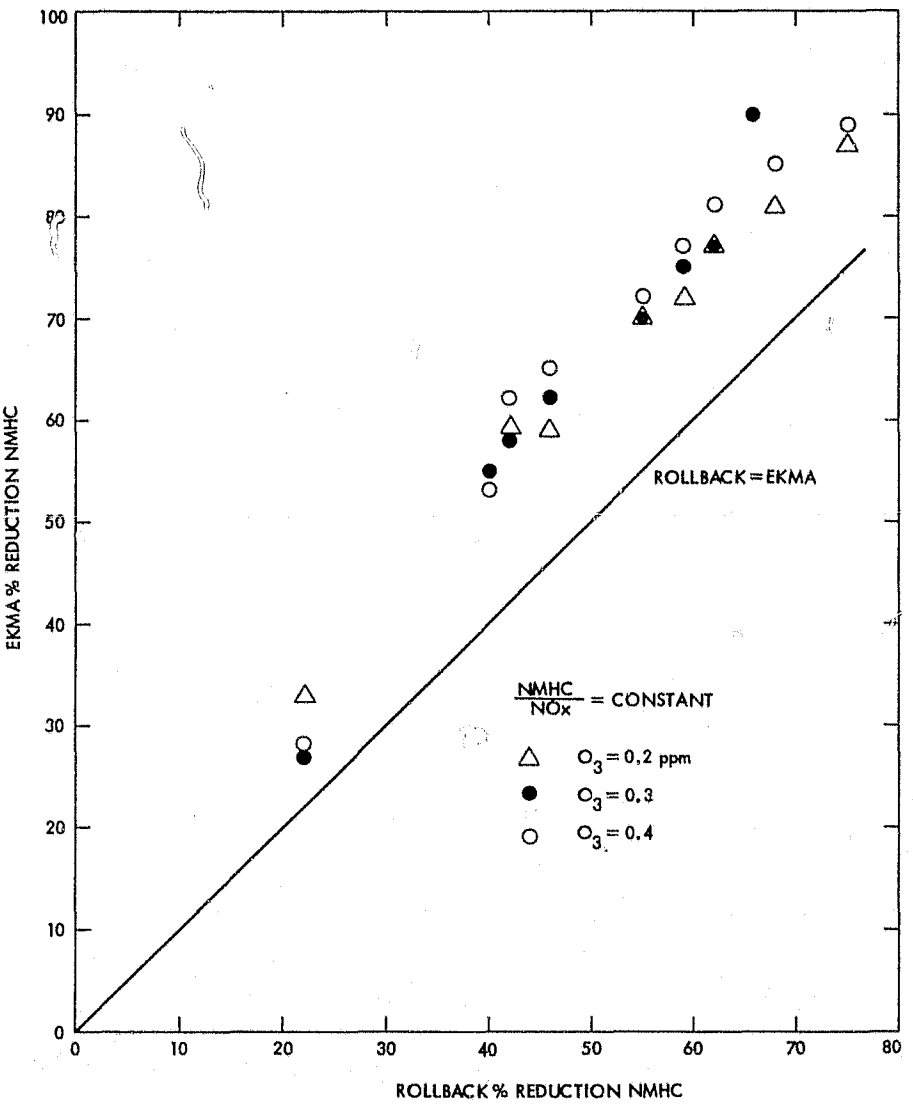


Fig. 4-5. Percentage Reduction Correlation Between EKMA and Linear Rollback Analyses of NMHC Holding $NMHC/NO_x$ Constant

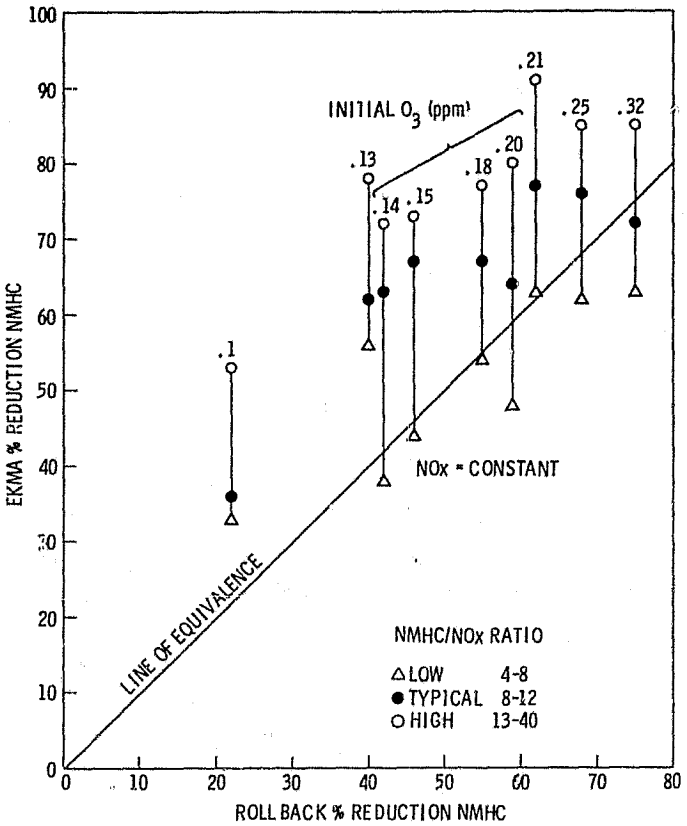
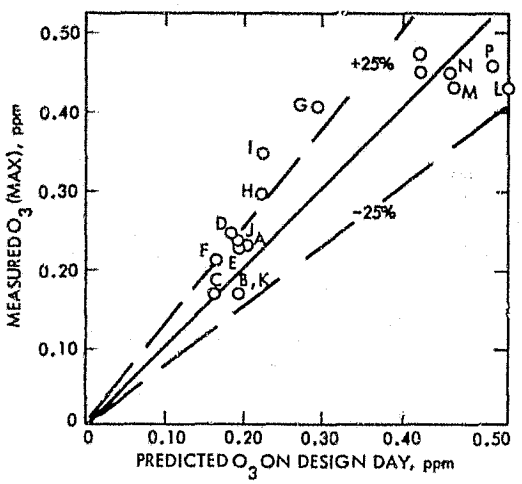


Fig. 4-6. Percentage Reduction Correlation Between EKMA and Linear Rollback Analyses of NMHC Holding NO_x Constant

Figure 4-6 has more detail than Fig. 4-5 since it indicates the initial ratio NMHC/NO_x, in addition to the initial concentrations of ozone. In this case for a low NMHC/NO_x ratio, i.e., 4 to 8, the EKMA predictions and linear rollback agree reasonably well, as may be seen by examining the triangular points shown in Fig. 4-6. However, as the NMHC/NO_x ratio is increased to the range of 8 to 12, one finds that the fit is not as good. In this range it appears that the EKMA reduction is approximately 20% higher than that attained on the basis of linear rollback. For high NMHC/NO_x ratios, i.e., 13 to 40, one finds that the EKMA predictions can be approximated by adding about 30% to the linear rollback predicted reductions. Hence, for both of the strategies of ozone reduction that have been examined, there appears to be a fairly simple relationship between the NMHC reductions based on linear rollback and those based on the Environmental Kinetic Modeling Approach.



A:	BAKERSFIELD	8/15/77	I:	SAN DIEGO	7/7/76
B:	FRESNO	9/3/76	J:	SAN JOSE/LOS GATOS	9/25/78
C:	SAN JOAQUIN	7/30/77	K:	S.F., RICHMOND/DELTA	9/25/78
D:	STANISLAUS	9/6/77	L:	LOS ANGELES/AZUSA	9/8/79
E:	SACRAMENTO	8/5/78	M:	LOS ANGELES/PASADENA	9/9/79
F:	SACRAMENTO	6/26/76	N:	LOS ANGELES/PASADENA	9/10/79
G:	LOS ANGELES/PASADENA	7/13/78	O:	LOS ANGELES/PASADENA	9/11/79
H:	ANAHEIM/RIVERSIDE	7/13/78	P:	LOS ANGELES/PASADENA	9/12/79

Fig. 4-7. EKMA versus Measured O_3 Concentrations

The California Air Resources Board has made a comparison of measured concentrations of ozone and those predicted using the EKMA approach. This data was supplied to our study by A. Ranzieri, Manager of the Air Quality Modeling Section of the California Air Resources Board (Ref. 4-13). This data is shown in Fig. 4-7, which is a plot of measured ozone vs predicted ozone for certain design days for 16 California regions. The cities and the design dates for each point are shown in Fig. 4-7. The figure shows that about 13 of the 16 points lie within slopes that are ±25% of the perfect slope, where the prediction agrees with the measured value. The result is surprisingly good in view of the stated accuracy that the Environmental Protection Agency attributes to the model of less than ±70% (Ref. 4-12, p. 72). Perhaps the Agency is being overly conservative in its estimation of the limitations on the Empirical Kinetic Modeling Approach.

In addition to the approach described above, the EKMA approach can be used to follow a given trajectory in an air quality region. Also, the chemical equations which are in the standard EKMA package can be modified if other equations are more appropriate for a given region. These modifications are described in the users manual (Ref. 4-14).

4.3.3 Photochemical Dispersion Models

The photochemical dispersion models are by far the most complex of the three under discussion. Excellent summaries of the current state-of-the-art in photochemical dispersion models are found in the reports by McRae et al. (Refs. 4-8, 4-15 and 4-16) and Reynolds et al. (Ref. 4-17). Earlier work by Eschenroeder et al. (Ref. 4-18) provides a historical perspective.

According to McRae et al. (Ref. 4-15),

> Modeling urban scale air pollution is essentially the problem of describing the formation and transport of chemically reacting species in the turbulent planetary boundary layer. The present model, based on the species conservation equation and a K-theory turbulence closure assumption, is given by
>
> $$\frac{\partial C_i}{\partial t} + \nabla \cdot (\mu C_i) = \nabla \cdot (K \cdot \nabla C_i) + R_i \ (C_i, \ldots, C_n, \ t) \qquad (1)$$
>
> where C_i is the concentration of species i, μ is the carrier fluid velocity, with components (U, V, W), and K the second-order, turbulent diffusion tensor. In current applications of the model chemical interactions R_i are described by the 50-step, lumped hydrocarbon reaction mechanism of Falls and Seinfeld (Ref. 4-19).
>
> Figure 4-8 is a simplified representation of various elements which either form a part of Eq. (1) or necessary inputs to the solution processes. The validity and accuracy of the various assumptions involved in the derivation of Eq. (1) have been discussed elsewhere (Ref. 4-20) and will not be repeated.

This model uses a three-dimensional wind field. A description of the procedures used to generate this wind field from the available ground data and data aloft is described in Ref. 4-21. The other components of meteorological factors in Figure 4-8 are cloud cover, radiation at ground level, temperature, inversion height, transport and turbulence, including the effects of surface topography. The pollutant sources in Fig. 4-8 are broken up into three categories: (1) anthropogenic sources, (2) geogenic sources, and (3) pollutants that have been transported into the region. Chemical processes are included in the model shown in Fig. 4-8. Homogeneous processes describe gas to particulate interactions. Various other processes for removal of pollutants are considered such as surface removal and other sink processes such as rainout. These components form part of the chemistry block. The information from the meteorological block, the source block, and the chemistry block enters into the mathematical block for calculation of the resultant concentrations of pollutants as a function of spatial coordinates and time.

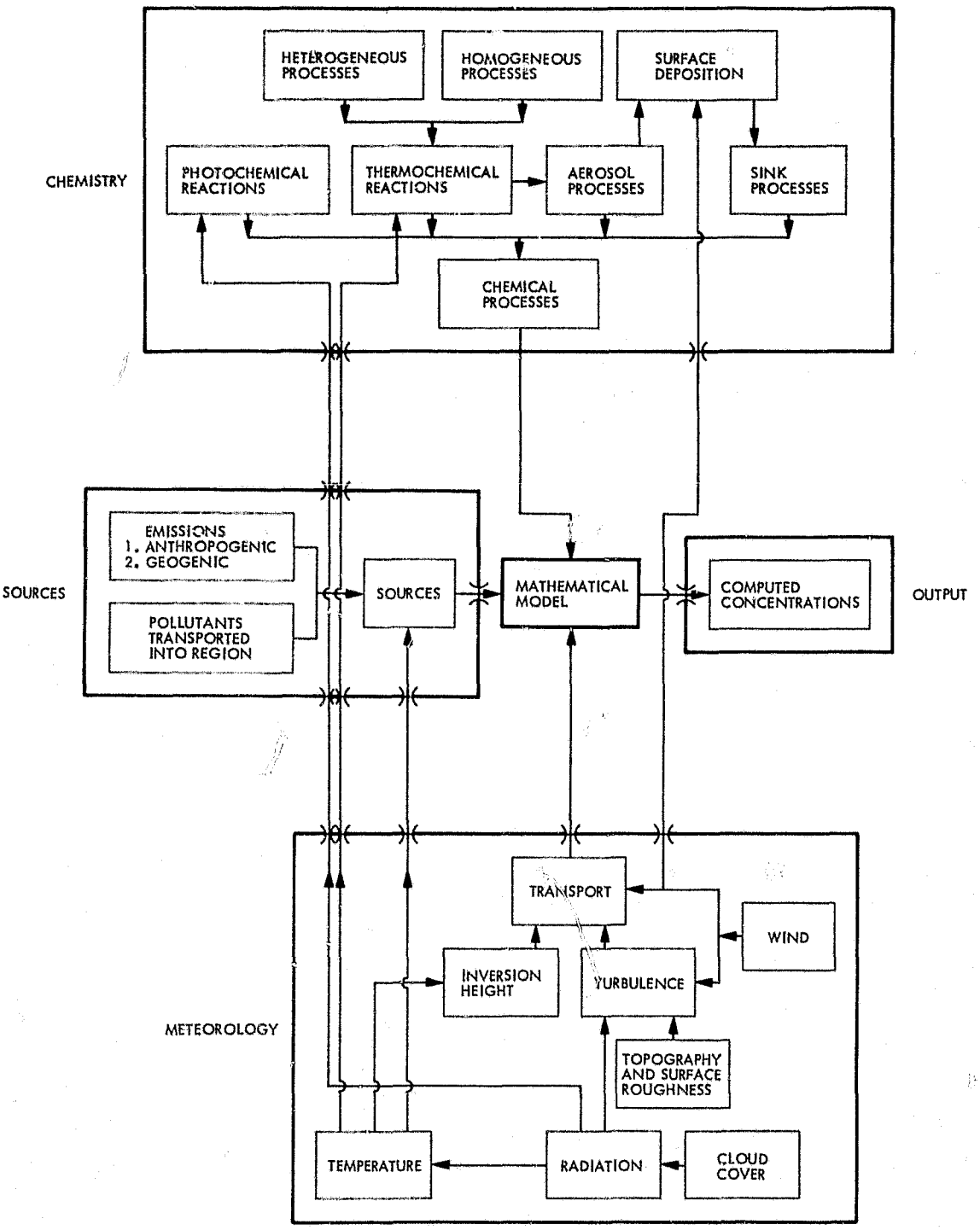


Fig. 4-8. Elements of the Photochemical Dispersion Model

This model requires that the emissions data be broken up into grid cells in accordance with the location of the emission sources. Typically the grid cells may be 5 km on a side. If a given grid cell has one or more major pollutant sources, these may be treated as a point sources, with their precise locations specified.

An example of the daily emissions inventory for reactive hydrocarbons for the South Coast Air Basin is shown in Fig. 4-9 (Ref. 4-21). The top part of this figure shows the five counties that are in the South Coast Air Basin: Ventura County, Los Angeles County, Orange County, San Bernardino County, and Riverside County. Below this county map are shown the total daily reactive hydrocarbon emissions for each grid cell. This map of emissions by grid square can be quite useful in determining errors within the emissions inventory of a given region. It is quite difficult to find the errors simply by going through the computer printouts because of both the basically dull nature of the work and the extremely large amount of data. However, a three-dimensional graphical presentation, such as that shown in Fig. 4-9, can cause certain errors to leap out. Hence, by putting the emissions inventory in this form, errors within the emissions inventory can be reduced. G. McRae reported that during his early checks of the emissions inventory for the South Coast Air Basin a coordinate error was somehow made. This caused an electrical power plant to apparently be emitting various pollutants out in the ocean! This kind of representation quickly shows up an error such as that.

This representation also shows the beginnings of the complexity of the use of a photochemical dispersion model for estimating the air quality in a region. It's quite expensive and complicated to obtain this much data, check it, and put it into such an emissions inventory. This data is not only needed on a 24 hourly average basis but is also needed on the time scale the order of an hour, in order to support predictions of the maximum hourly concentration of ozone. R. Blackwell of the Jet Propulsion Laboratory and G. McRae of Caltech's Environmental Quality Laboratory produced a set of non-methane hydrocarbon emission isopleths for each hour on June 26, 1976, to demonstrate the hourly variability of the emissions of reactive hydrocarbons. These isopleths are shown in Fig. 4-10. An examination of these patterns for the hours of 2:00 a.m., 3:00 a.m., 4:00 a.m., and 5:00 a.m. reveals very little change in the emission patterns. However at 6:00 a.m. the influence of daily commuting back and forth to work starts to considerably change the emission patterns. Major increases tend to follow the freeways and also occur in the major cities. For example, the reactive hydrocarbon emissions in terms of kilograms per hour per grid cell, in the area of Pasadena and Los Angeles, are taken from Fig. 4-10 and displayed in Table 4-1. Pasadena is clearly identified in the maps in Fig. 4-10. In order to determine the location of Los Angeles, for example, consider the set of isopleths at 9:00 a.m. Just to the west of Pasadena there is a very bright white spot shown corresponding to greater than 1,000 kg per hour per grid cell. This spot represents downtown Los Angeles.

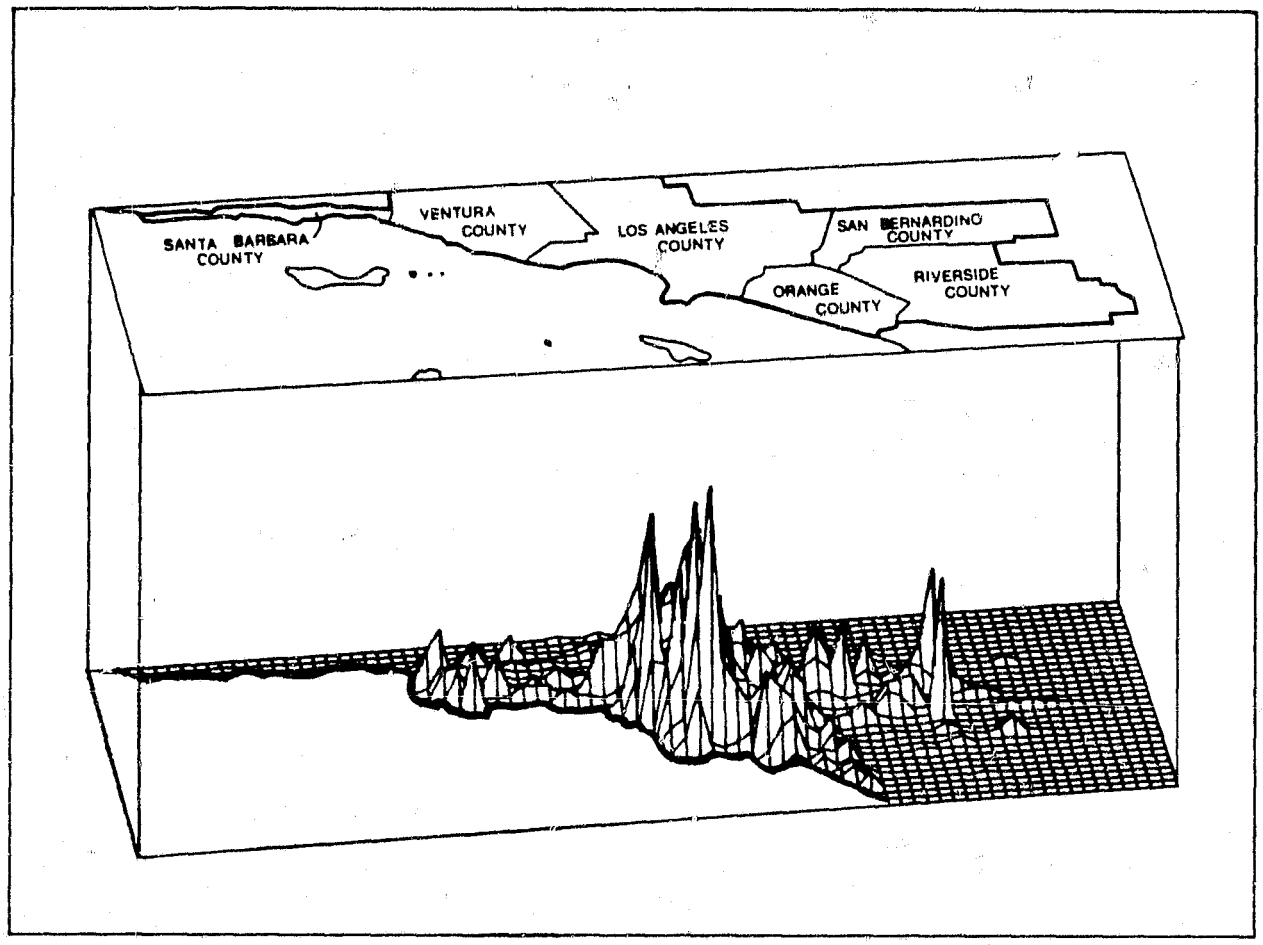


Fig. 4-9. Reactive HC Daily Total Emissions

At between 2:00 in the morning and 5:00 in the morning the emissions in both Pasadena and Los Angeles are quite constant. Pasadena's range is in the order of 21 to 60 and Los Angeles from 61 to 150. In both areas the emissions start to change dramatically at 6:00 a.m. By 8:00 a.m. Pasadena has peaked at an emissions between 601 and 700, whereas Los Angeles is still increasing in the range of 831 to 999. The emissions drop somewhat in both areas between the hours of 10:00 a.m. and about 3:30 p.m. For example, Pasadena has values ranging between 451 and 600 from 9:00 a.m. till 5:00 p.m. At 5:00 in the afternoon the Pasadena value increases to reflect the increase in traffic. A similar phenomenon occurs in Los Angeles at 4:00 p.m. and continues until 7:00 p.m. By midnight in both areas the emissions have approached the 2:00 p.m. emissions of the same day. Similar tables could be constructed for all the major cities in the South Coast Air Basin. Hence, one can see that the representation in Fig. 4-10 contains an enormous amount of data coded in such a way as to be highly visible to allow the modeler or the air quality planner to see quite easily what changes are taking place on an hourly basis.

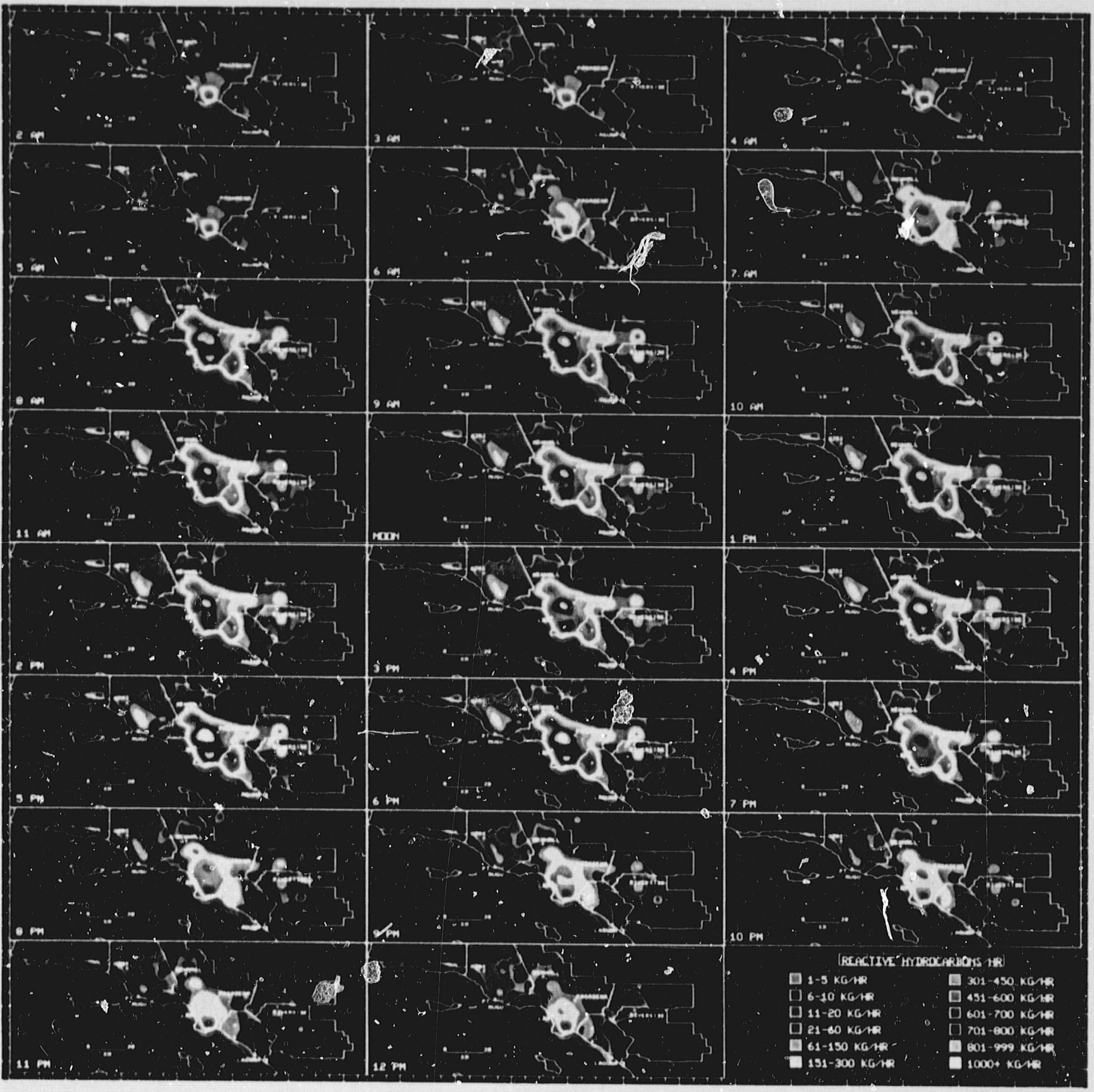


Fig. 4-10. Hourly Non-Methane (Reactive) Hydrocarbon Emission Isopleths for the South Coast Air Basin

In any kind of modeling activity of this level of complexity, there is danger that errors will combine in such a way as to limit the usefulness of the data. A discussion of errors in photochemical dispersion modeling was provided by John Seinfeld in Ref. 4-22. The essence of the discussion follows:

If we summarize uncertainties...we have:

1. Chemical kinetics (ozone levels)	±50%
Initial concentrations (HNO_2, aldehydes)	
Rate constants	
Mechanisms of hydrocarbon oxidation	
2. Meteorology	
Wind speeds and direction	±20
Mixing depth	±25
Light intensity	±20
3. Initial and boundary conditions	
Initial concentrations aloft	±50
Boundary concentrations aloft	
4. Emissions inventories	
NO_x	±20
Hydrocarbons	±30

> In the absence of detailed accuracy evaluation studies, an estimate of the uncertainty in predicted ozone levels as a result of the above input uncertainties acting individually or in concert represents sheer guesswork... We have singled out two of the most influential uncertainties, namely initial concentration of radical-producing species and the upper level boundary conditions, together with the question of the degree of vertical resolution for more detailed study. On the basis of [this] ... and the prior validation studies cited in the previous section, one is inclined to place an overall uncertainty on oxidant level predictions from current AQSM [air quality simulation models] of ±50 percent. We therefore conclude that: Oxidant level predictions of current grid-based AQSM (i.e., the SAI and LIRAQ models) have an estimated uncertainty of 50 percent.

4.3.4 Air Quality Model Calibration and Validation

Various approaches can be used to calibrate and validate air quality models. Simple models such as the linear drawback model can be calibrated by the technique described in Section 4.4.1. That is, simply take a given year where the air quality is known and the emissions are known. This then calibrates the model for the years. A more sophisticated approach can be used for linear rollback and calibration if many years of air quality data are available. For a given calibration year a plot of the rollback predicted values for other years vs the measured values can be made. This plot can be fitted by means of a linear regression straight line, giving a correlation coefficient and a zero offset. This process can be continued for several years, each time choosing a different year as the basis of calibration. One can then simply pick the year as the optimal calibration year which produces the highest correlation coefficient and smallest zero offset. This procedure would then produce the best values of background B and

α shown in Section 4.4.1. That is the emissions proportionality factor.

Table 4-1. Reactive Hydrocarbon Emissions Rate for Grid Cells Centered in Pasadena and Los Angeles

Time	Grid Cell kg/hr	
	Pasadena	Los Angeles
2 am	21-60	61-150
3 am	↓	↓
4 am		
5 am		
6 am	61-150	151-300
7 am	301-450	451-600
8 am	601-700	801-999
9 am	451-600	1000 +
10 am	↓	801-999
11 am		↓
12 am		
1 pm		
2 pm		
3 pm		
4 pm		1000 +
5 pm	601-700	↓
6 pm		
7 pm	301-350	701-800
8 pm	151-300	451-600
9 pm	↓	301-450
10 pm		↓
11 pm	61-150	151-300
12 pm		

The EKMA model can be calibrated using the techniques described in the users' manual. We shall not describe these in detail since many of the issues involved in this kind of calibration are similar to the issues involved in the photochemical dispersion model calibration, which will be discussed next.

Alternate approaches can be used to calibrate photochemical dispersion models. In one approach the model can be considered to be self-calibrated; that is, the information that is put into the chemical models and the meteorological models and the emissions models is considered to be the best information available and these values are simply not adjusted. Hence, the model tends to act as if it were producing absolute values. This is the approach used in the Caltech photochemical dispersion model activity. In contrast, an approach can be used in which certain parameters are adjusted to calibrate the model to the region. For example, in the application of the APRAC II Carbon Monoxide Model to the Phoenix Air Quality Region, it was necessary to both choose an inversion height of approximately 15 meters and multiply the predicted CO concentrations by a factor of 2.6 (Ref. 4-23) in order to get a good agreement between the time-dependent concentrations at the stations and those predicted by the model. The following indicates the reasoning accompanying such calibrations:

> The standard error of the adjusted estimate is a .9 ppm which represents an eight percent standard error. The linear relationship between raw and adjusted APRAC estimates and observed concentrations is illustrated in Fig. IV-6 (see Ref. 4-23).
>
> Although there were only five sites monitoring CO on January 16-17, 1975, the correlation between the observed and adjusted estimates (Fig. IV-6, Ref. 4-23) is significant. This fit indicates that APRAC-II is able to simulate the spatial variation in eight-hour CO concentrations under typically severe conditions within reasonable bounds of accuracy.
>
> One explanation for the large correction factor and low mixing height required to achieve accurate absolute estimates with APRAC-II is the high "background" level of CO actually present at the start of the modeling period, i.e., 8 p.m. This "background" level is caused by the p.m. peak traffic emissions trapped at sunset by ground-based inversion conditions. The correction factor and 15-meter mixing height are methods of artificially creating those emissions trapped between 4 and 8 p.m. An alternative method would be to apply APRAC for a 12-hour period beginning at 1600 on January 16, 1975. The latter is a preferable approach theoretically but increases APRAC computer costs by 50 percent. For this reason and prevailing time constraints, the factoring approach was adopted as the calibration method for the AQMP reanalysis. However, given increased resources and the use of APRAC-II in future regional CO analyses for Phoenix, the 12-hour averaging approach should be investigated as an alternative, and perhaps more accurate, simulation technique.

The above is an example of using a calibration technique.

If there are many parameters that can be adjusted on calibrating the model, then the model becomes of questionable value simply because its results can be interpreted as a form of N-dimensional space curve fitting. Hence, the more absolute approach that is used in the Caltech modeling would be intrinsically preferred.

However, in the use of any model there may be some subjective manipulation of the data which is not done in an intentional manner. For example, the emissions inventory data has errors in it. These errors may be adjusted in such as way as to cause the model to give a better fit. If the model gives a bad fit on a given day, this may be indicative to the modeler that there is some problem within the emissions inventory. He may then attempt to correct this problem, run the model again, and find out if the model gets better agreement or not. If the agreement is better, then it is assumed that the initial problem was due to the fact that the emissions inventory had a certain category of error. Other errors in the emissions inventory may actually help the model. Hence it is difficult to determine the preciseness of the model when this approach is used. It is suggested that another approach be used to determine the intrinsic value and variability associated with any given air quality simulation model.

If, for example, modelers were allowed to adjust their models with whatever calibration techniques they saw fit, for perhaps one or two days, then the coefficients within the model would all be fixed and no longer changed. If the emissions inventory for perhaps 30 days could be then put into the model by other modelers who did not have a direct interest in the outcome of the test, one could then make a comparison of the actual output of the model without the model being changed in any way for 30 different days of emissions inventory data. In this way one could obtain a good indication of both the accuracy and the precision of the given air quality simulation model and also avoid some of the potential biases that occur in having a given modeler decide on the accuracy and validity of his own model.

This approach would be extremely expensive and time consuming. It has been estimated for areas such as the South Coast Air Basin that to prepare an emission inventory for a single day would cost in excess of $30,000. If 30 consecutive days of emissions inventory were prepared it would be expected that there would be some deficiencies of scale obtained and that the cost would not be simply 30 times $30,000. Perhaps the overall cost might be reduced to the order of 30 times $15,000 or $450,000. This cost would only be the first cost in a series of costs. Once the emissions inventories were prepared on a daily basis and the other factors were known, (such as the meteorological and possible chemical factors that might enter into the model), the model would have to be run. If it cost on the order of $1000 for a given day, this type of model run would then cost an additional $30,000. These numbers may be on the optimistic side. In addition, it might cost an additional $30,000 to calibrate the models. Then an evaluation of the results of the model as compared to the observed air quality data would have to be made. This evaluation might cost an additional $200,000. Hence, the overall effort for calibration and validation of a model for an area such as the South Coast Air Basin

in this manner would cost on the order of three quarters of a million dollars. This price tag may seem awfully high, although it should be noted that similar efforts at producing a data base for model evaluation conducted by the Environmental Protection Agency in the regional air pollution study at St. Louis cost approximately $25,000,000. So perhaps three quarters of a million dollars is not too high a price to pay for evaluating an air quality simulation model in an area such as the South Coast Air Basin.

Other approaches to understanding the limits of the air quality simulation models can be used. An extensive set of information (Ref. 4-24) is available on the sensitivity of the Caltech photochemical dispersion model for changes in various chemical parameters. A Monte Carlo approach similar to that shown in Fig. 4-11 could be used in determining the overall sensitivity of a complex model to variations in the input parameters to the model. Each input parameter for meteorological factors, source emission factors, or chemistry has a best value estimate. Associated with each best value is an estimate of the precision of that value, that is, the error involved in the best estimate. If the assumption were made that the errors were distributed in a gaussian manner, then each parameter could be characterized by a best estimate and a standard deviation. In this case, the highest probability for a given parameter is the best estimate. Other possible values of the parameter have lower probabilities as expressed by a gaussian bell-shaped curve. Other than gaussian probability distribution functions may be appropriate for certain parameters.

In the Monte Carlo process, the computer could randomly select a value from the gaussian curve for each parameter. In the meteorological area, for example, one value each could be picked for inversion height, turbulence, wind and cloud cover. Similar random choices would be made by the computer in the areas of emission inventory and chemistry. Thus, for a given run, the computer would randomly pick from the gaussian curves a set of values for all parameters, and then calculate the predicted concentrations of the air pollutants of interest. A second run would then be performed in which the computer would repeat the picking process, selecting a different set of parameter values from the distribution curves. Once again, a computed set of air pollutant concentrations would be generated. This process would be repeated 40 to 60 times in order to get information on which output concentrations occur more frequently than others. Hence, for a given location, one could make a plot of computed concentration versus the number of times it had occurred, such as that shown in Fig. 4-11. Using techniques such as this, one can arrive at an unbiased estimate of the effect of all the uncertainties within air quality simulation models. This also provides an unbiased estimate of both precision and accuracy of air quality simulation models.

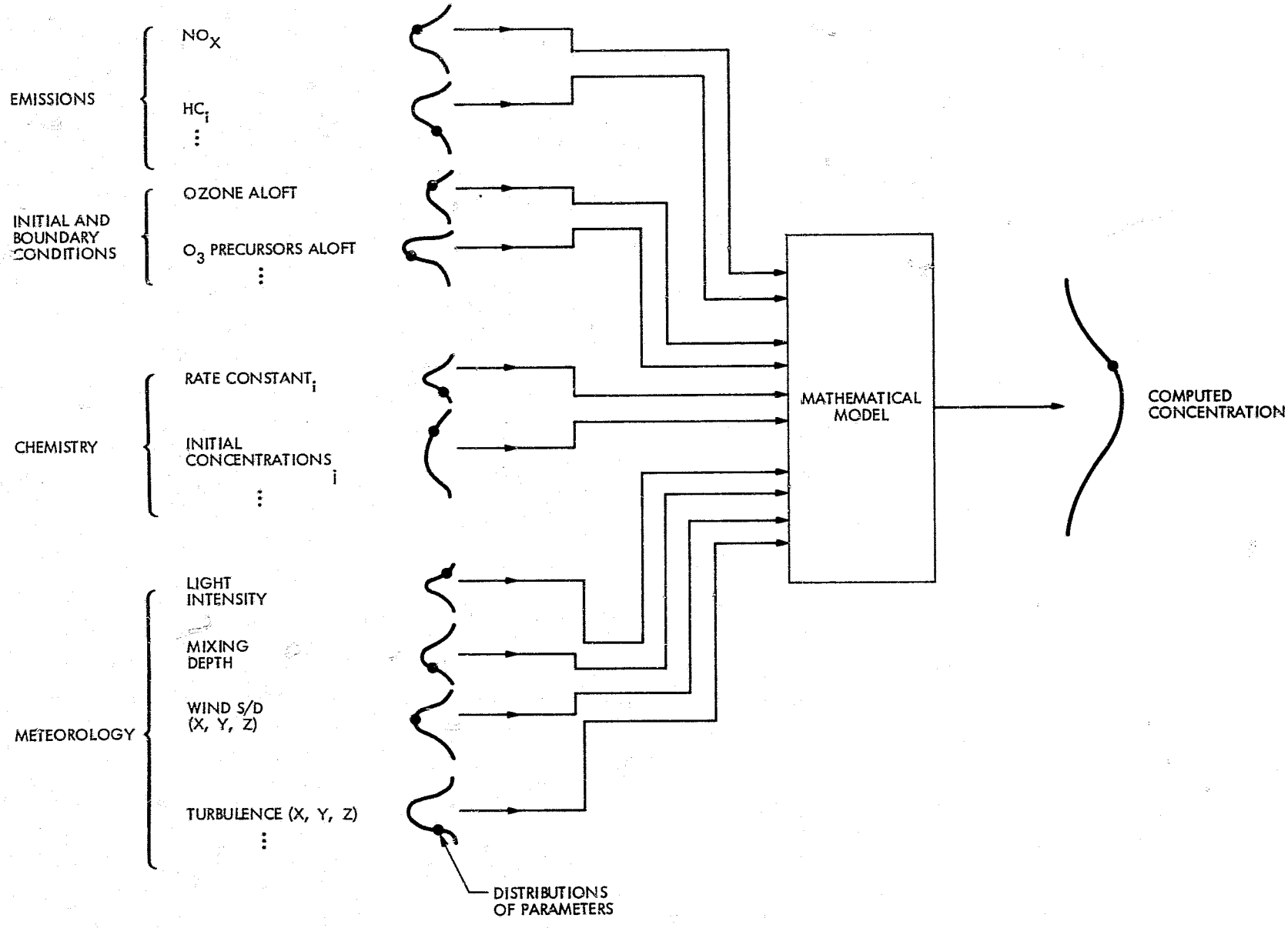


Fig. 4-11. Monte Carlo Simulation of Air Quality to Estimate Uncertainties Based on Statistical Distributions of Parameters

The cost of performing such estimates would be far less than the cost of the test described earlier. It would require a good set of emissions inventories for perhaps 2 or 3 days. These emissions inventories may already be available. The estimates of precision would require some rationale for determining the appropriate error in each parameter going into the model. This undoubtedly would be the most expensive part of the precision check for an air quality simulation model. In many cases the error in the data will simply not be known. It may be that some sort of a delphi technique might be used to determine the estimates of the errors in each parameter going into the model. This approach could be used with a separate set of errors for each parameter. Let's make a rough estimate that determining the errors for each parameter might cost on the order of $200,000. Some modification would have to be made to the standard air quality simulation model to add the Monte Carlo function to it. It is assumed that this modification would be fairly inexpensive: perhaps $20,000. Then the model would have to be run 40 times. Assuming that each run of the model costs $1000, this might amount to an additional $40,000. After these runs were made, an analysis of the data would have to be prepared. This analysis might cost $30,000. Hence, in order to determine the precision and accuracy of a given air quality simulation model for a given set of emissions inventory data, meteorological data, and chemistry, the cost would be approximately $300,000.

Approaches other than the Monte Carlo approach described above might be used to estimate the precision of this technique. The Monte Carlo approach is mentioned because it has been used extensively to analyze complex systems such as computer systems, electronic circuits and transport and dispersion in nuclear systems. Hence, it is a conservative approach to determining the precision of an air quality simulation model. The authors are not aware of this process having been done on any air quality simulation models where the effects of uncertainties in emission inventories, meteorology, atmospheric chemistry and measurements have been treated simultaneously.

4.3.5 Air Quality Model Issues

4.3.5.1 Model Complexity

One of the principal issues facing an air quality manager is the degree of sophistication that should be applied to air quality modeling. In this report we have seen three air quality models that vary quite widely in sophistication, ranging from the simplest, being the linear rollback model, to the most complex photochemical dispersion class of models. The choice of a particular model would depend on many factors. Some of those factors will be considered in this section. In an article entitled "Sensitivity and Sensibility in Air Quality Models" (Ref. 4-25), M.L. Hatmaker makes many interesting points.

> The sophistication of atmospheric pollutant emissions and diffusion models (and data requirements) is advancing rapidly. On the other hand, the quality of the basic input data and the practice of traffic simulation modeling are not receiving the same level of attention. The air quality models operate using the output of the traffic simulation process demanding data for which the accuracy is difficult to confirm. The dangers of proceeding by chains with sub-models which have specification error using data which are subject to measurement error are apparent. It would be desirable to use sophisticated air quality models to analyze the effectiveness of a set of Transportation Control Measures (TCM) in reducing pollutant concentrations. However before these costly simulations are undertaken- less sophisticated, aggregate approaches should be used to examine the nature of the problem. These analyses may provide enough information to justify the traffic, emissions, and diffusion simulation process. In some cases, this information may indicate that it would be a futile effort to attempt to use the full scale simulation process to model the TCM strategy. In these cases a less demanding method may be appropriate and give useful results.

This same type of message has been made previously by W. Alonso in Ref. 4-26. Alonso sets forth the fundamentals as follows:

> Long chains of argument are the delight of theorists in the source of their mistrust by practical men. There is some merit in this distrust. Imagine that we argue that if A then B, if B then C, etc. If we are 80% certain of each step in the chain, from the joint probability of the steps it follows that we are less than 50% certain of where we stand after four steps... In this paper I will raise the issue of the effects of errors and their propagation in models for prediction, and suggest some strategies for the selection and construction of models which are intended for applied work. The gist of my argument is that the use of sophisticated models is not always best in applied work, and that the design of the model must take into account the accuracy of the data on which it will be run. There exists the possibility, which should be explored, that some of our most intellectually satisfying models should be pursued as fundamental scientific research, but that simpler and more robust models should be used in practice.

Alonso makes the point that it is quite important to distinguish between two types of error: error of specification and error of measurement. Errors of measurement are described in some detail in Section 5 of this report, in particular, measurement errors involving the oxides of nitrogen, nitrogen dioxide and nitric-oxide, non-methane hydrocarbon measurements, ozone measurements, and the implications these have to the ratio of non-methane hydrocarbon to NO_x. Since errors of measurement are treated in some detail in Section 5, they will not be discussed further in this section.

Hence, we will confine our attention to errors of specification. In considering the three types of air quality simulation models that we have discussed thus far, linear rollback certainly has the most errors of specification. Its application is of questionable validity in terms of the physical basis for the theory. The environmental kinetic modeling approach (EKMA) is an improvement in this regard in that it recognizes the behavior of both non-methane hydrocarbons and NO_x in the air in terms of their contributions to the formation of ozone. However, it also leaves much to be desired in that it artifically lumps the NO_x measurements and the non-methane hydrocarbon measurements into some average between 6:00 and 9:00 a.m. in order to predict ozone peaks later in the day. In addition, the EKMA approach considers only certain trajectories within an air basin and does not consider the entire air basin. Hence, we can eliminate all of the above objections by going to the photochemical dispersion model. Of the three considered, this model has very small errors of specification. Hence the errors of specification decrease as the level of sophistication in the model increases, and correspondingly, the number of parameters that go into the model increases.

This approach is not without its drawbacks, however, since the number of parameters in the photochemical dispersion diffusion model is quite large, and the measurement data base upon which analyses are made becomes sparser and sparser. For example, in the 29-species set of reactions that are used in the Caltech photochemical dispersion model, only three of the species are pollutants that are regulated by the Environmental Protection Agency. Hence, extensive measurements are only available for those three pollutants. Rather more meager data bases are available for the other 26 pollutants that are in the model. Hence, the errors associated with using the other 26 pollutants individually can be significantly higher than the errors associated with using the regulated pollutants. This is an example of a situation in which the specification error becomes very small by very closely specifying the chemistry of the situation. However, the measurements error can become considerably larger because there are measurements of the species used in the chemical reactions.

Another example of this kind of improvement in specification but worsening in terms of measurement errors involves the meteorological data. For example, referring back to Fig. 4-8 in this section, cloud cover may not be well known throughout the entire basin, so some estimate of cloud cover may be made from one station, or estimates may be made by using a photometer. The temperature at the surface of the basin may be well known by the air monitoring stations and data from airports. However, the temperature as a function of altitude is, in general, rather poorly known except for two times during the day. Our knowledge of the behavior of temperature vs altitude in between those times is quite meager. Hence, once again by specifying the system in greater detail, we may increase the errors in measurement.

Similar comments could be made in terms of the wind field. We have a rather good set of information concerning the wind distribution on the surface, again from the air quality monitoring stations and the stations which measure horizontal wind speed and direction at the airports. However, our knowledge of wind aloft is quite meager except for the measurements which are made at the airports twice a day. Thus, phenomena such as wind sheer, where the wind direction will suddenly change between two layers of air, are not included in the model. Interesting phenomena can be seen caused by wind sheer in observing the plumes from two neary smoke stacks. One smoke stack might be such that its plume gets carried to the right, whereas the second plume might have a higher plume rise and get into a layer where the wind is sheered and this plume might be sheered to move to the left. Hence, the smoke coming from two plumes could, in certain situations, be going in directions 180 degrees apart. Again by specifying the wind field and with very limited measurements of wind aloft, we run into the danger of increasing the error of measurement by decreasing the error in specification. Similar comments can be made for the emission inventories. Estimates of hourly emission rates are made from some data which is, at best, credible on a yearly basis. Other emission factors are used which are highly variable in themselves.

A general theoretical approach to summarize this type of situation has been put together by Alonso (Ref. 4-26, p. 184). Fig. 4-12 is a graph of total system error on the vertical axis vs system complexity on the horizontal axis. As was mentioned earlier, the error in specification E_s decreases significantly as the complexity of the system increases. However, for a given set of measurements, the error of measurements increases significantly as the complexity of the system increases. Thus, specification and error in measurements tend to go in opposite directions as the system complexity increases. That is, the error in specification decreases and the error in measurements increases, simply because we have limited measurements of some of the new materials that are specified. In Fig. 4-12, all the errors of specifications have hypothetically been incorporated into the curve marked E_s; all the errors of measurement are included in the curve marked E_m. Hence, the total error in the system is simply obtained by adding the errors of specification to the errors of measurement for a given level of complexity. His top curve, which is marked E, is the total error in the system as a function of system complexity. It should be noted that this curve of total error has a minimum which is reached for levels of complexity that are small compared to those to the far right of his complexity scale.

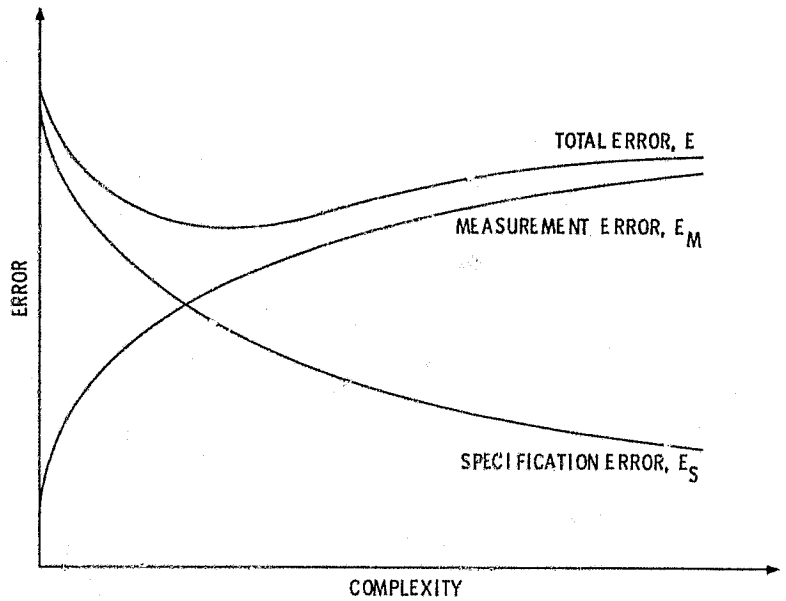


Fig. 4-12. System Error as a Function of Complexity

Alonso extends this concept to two sets of measurements, E_m^* and E_m, shown in Fig. 4-13. In the first set, E_m^*, the errors of measurement are larger than those of a second set, E_m. This difference could be due to improvements in factors such as instrumentation, quality control, training, etc. Since the measurement errors E_m are less than those of E_m^*, this has the effect of driving the minimum error as a function of complexity farther to the right side of the graph, i.e., from point A to point B. As the measurement error decreases, the overall minimum system error tends to lie toward more complexity. Hence, by increasing the complexity of the system we can decrease the total error in the system, if the measurements accuracy justifies this change. On the other hand, if the system complexity is greater than the optimum value for a given set of measurement error, then increased complexity simply increases the total system error.

The authors are not aware of any analysis of this type that has been done for a complete air quality simulation model involving meteorological factors, emission inventory factors, and chemistry reactions in addition to the intrinsic uncertainties that may be in the mathematical model. It appears highly desirable that this type of analysis be performed in order to allow estimates to be made of the appropriate level of complexity in terms of minimizing the error in the overall system analysis. It should be noted that in practical air quality management this type of discussion only incorporates one rather limited set of variables. Concepts such as the amount of money available

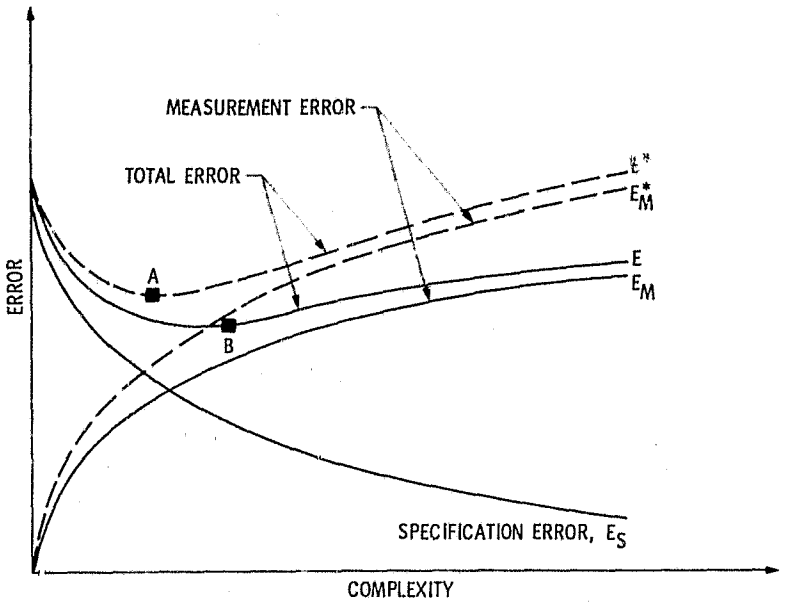


Fig. 4-13. System Error versus Complexity for Two Sets of Measurements Errors

to a given region, the number of people, and the levels of sophistication are not included in this analysis. These kinds of considerations may tend to drive the overall analysis toward even less complexity than that which would be indicated on the basis of the analysis suggested above.

4.3.5.2 Spatial Dependence

The spatial resolution of an air quality simulation model will depend on many of the factors within the model. One factor which gives an obvious dependence is that of wind speed. For example, if the wind speed in an area for a given hour is 5 miles per hour, then a parcel of air will move 5 miles in one hour during that hour. The actual movement of the parcel of air will of course depend on the detailed direction of the wind speed and its speed as a function of time within that hour. For example, if during half of the hour the wind is blowing in the northeast direction and for half of the hour it blows in the southeast direction, the simple averaging concept would assume that the wind during this entire hour had blown in the east direction. If the plume under consideration interacts chemically with other chemicals that may be coming from point sources, these concentrations could be influenced dramatically, depending on whether the plume was blowing in the northeast direction or in the southeast direction. Hence we have an intrinsic error in terms of the chemical interactions that take place.

According to Slade (Ref. 4-27), "In the first thousand or so meters above the earth's surface, the wind speed and direction are determined primarily by three forces: the force due to the horizontal pressure gradient, the Coriolis force due to the earth's rotation, and the frictional force due to the roughness of the earth's surface."

In making estimates of the behavior of the wind during interpolations from the hourly average data, the inclusion of parameters such as the horizontal pressure gradients may tend to increase the validity of the interpolations. However, it should be noted that an extensive amount of averaging has already been applied to the data and there is no way to reconstruct what the wind directions actually were, other than going back to the original wind data which is taken on perhaps a second by second basis with time average ranging from 1 to 10 seconds. Thus, going back to the example cited earlier, if the wind direction changes by 90 degrees over a period of one hour, one finds that the net wind direction for the hour is approximately that given by two vectors, one going northeast, the other going southeast; that is, a single vector going in the easterly direction. Since it is measured on a separate sensor, the effective wind speed would indicate 10 miles per hour rather than 7 miles per hour, which would be the effective wind speed going in the easterly direction. Hence, the degree of fluctuation of the wind places intrinsic limits on the spatial dependence that can be accurately represented in an air quality simulation model.

4.3.5.3 Temporal Dependence

Similar comments can be made about the temporal dependence in addition to the uncertainties involved in the averaging processes due to averaging wind speed. Other uncertainties are involved in the kinetics of the chemical reactions that take place. These in turn effect the real temporal resolution of the model. The models that are used should be able to have a temporal resolution consistent with the primary ambient air quality standards being modeled. In the case of NO_2, the standard requires an averaging of one year. In the case of ozone, the temporal dependence must be reasonably accurate on the basis of a single hour. Since NO and NO_2 are precursors of O_3, it is necessary to model them on an hourly basis, if a region has an O_3 air quality problem.

4.3.5.4 Multiple Day Interactions

As was pointed out in Section 4.1.4, multiple day meteorological effects are quite important in some air quality simulation models, especially those which involve predictions of hourly average concentrations. It should be noted that at the Photochemical Modeling Workshop of the National Commission on Air Quality one of the issues presented was the basic lack of data concerning the rates of chemical reactions at night. Hence in doing a multiple day simulation it would

C-2

be very important to adequately validate whatever information is available on the combined effects of chemistry and meteorology during the nighttime hours. Information is needed on both homogeneous interactions and heterogeneous interactions. If this data base remains unvalidated it will be difficult to credibly extend the simulations meaningfully beyond a single day. It has been pointed out previously that the boundary conditions, that is, the initial conditions of concentration of a variety of pollutants, can be very significant in determining air quality simulation predictions for a given day. It appears that much more work is needed to enable one to be able to accurately describe what is occurring in pollutants that are aloft during the night. This importance extends not only to a city such as Los Angeles in which the wind may slosh back and forth such that in effect it is "downwind of itself," but this could also have impact on cities which are downwind of other cities.

4.3.5.5 Background

Influence of background has been discussed previously in Section 4.1.5. Once again this influence is a major issue in present air quality models. Not only is background from such exotic sources as the stratospheric intrusion of ozone important, but also background from other more normal sources such as hydrocarbons that are emitted both from the ground and the trees could be important.

4.3.5.6 Natural Sources

At the Workshop of the National Commission on Air Quality there appeared to be considerable variation in the emission factors associated with hydrocarbon emissions from vegetation. It was stated that these emission factors could easily be off by 4 orders of magnitude. It appears that there is a need to pin these emission factors down much more certainly so that one can decide the true significance of natural sources as compared to anthropogenic sources. This type of decision could have a very large impact on air quality management. At present the data is apparently unclear, which causes a polarization in air quality management. Some feel that natural sources contribute very strongly to the degradation of air quality in a given region, whereas others feel that natural sources make only a small contribution. It appears vital to remove or at least minimize this uncertainty so as to reduce the polarization that has occurred in the political process.

4.3.5.7 Limitations Due to Type of Application

In the use of air quality simulation models there is need for sufficient time, money and skills. If these parameters are not available in adequate amounts then large compromises may be in order. The 1977 amendments to the Clean Air Act do not appear to have allotted sufficient time, dollars or skill for the first attainment plans. It is hoped that the process is viewed as an organic process in which skills develop on a year-by-year basis and hence quality of the plans,

both the local plans and the state plans, should increase from year to year. The requirements for a reasonable further progress report should tend to drive the system in this direction. Those regions which are nonpayment regions for one or more primary pollutants in principle will have better plans submitted for the 1982 SIP than for the SIP required in 1979. However, it should be noted that there may be still fundamental uncertainties that have not been resolved in the planning process. These uncertainties may need to be defined by the Congress in the proposed 1981 amendments to the Clean Air Act in order to insure that they are incorporated properly by local regions. In order to insure this incorporation it would be essential for the Congress to provide adequate funding to staff the problem at the appropriate level and specify more clearly milestones which would be obeyed not only by the air quality management districts but also by the Environmental Protection Agency in specifying rules and regulations for local areas. Timely milestones should also be specified for the states such that the process is clearly understood at the three levels: local, state, and federal. The current process incorporates continuous changes at these three levels which serve to frustrate and in some cases cause deterioration in the quality of the analysis that is being performed.

4.3.5.8 Chemical Reactivity Scale for Non-Methane Hydrocarbons

At present the EKMA type modeling that the Environmental Protection Agency calls for uses the concept of non-methane hydrocarbons. The concept is based on the idea that a class of hydrocarbons reacts so slowly that essentially it does not take part in the photochemical process. Hence, this class of hydrocarbons, that is "non-reactive hydrocarbons," is eliminated from the modeling activities. It should be noted that the reactivity of the reactive hydrocarbons can vary over several orders of magnitude for a variety of reactive hydrocarbons," that are found in urban atmospheres.

Hence, it may be very important to develop emission inventories on the basis of classes of reactivity. Furthermore, if air quality management continues along the line of simply measuring total reactive hydrocarbons, a decrease in the daily emissions of one kind of reactive hydrocarbon may be attained by shifting to a more reactive hydrocarbon. If the quantity of hydrocarbons is decreased by, for example, a factor of 2, this might look like improvement. However, if the reactivity is increased by a factor of 10, which could easily happen, then the net effect would be to worsen the effective emissions from that source by a factor of 5. This issue appears to be one of great importance and should be resolved in order to prevent control tactics which are counterproductive in terms of their influence on photochemical smog.

4.3.5.9 Modeling Uncertainty

In this section we summarize the uncertainty associated with the air quality models. Earlier an estimate was given for photochemical dispersion models of +50%. This estimate was based on the assumption that the emission inventories are in relatively good shape; that is, that the errors associated with the emission inventories are small compared to the errors associated with the intrinsic behavior of the photochemical dispersion model. The number of ±50%, then, does not really reflect the errors that could be associated with large percentage errors in emissions inventory. A similar estimate was made for the empirical kinetic modeling approach (EKMA). This estimate was ±70%. In the section where EKMA was compared with the linear rollback model, it was shown that EKMA tended to be biased upward in terms of its emissions reductions in non-methane hydrocarbons. Referring back to Fig. 4-6, for example, for low NMHC/NO_x ratios the EKMA model agreed quite closely with the linear rollback model. For higher NMHC/NO_x ratios the EKMA model was biased somewhat upward in terms of reductions compared to linear rollback. Hence, we estimate that the lower limit of the rollback error is comparable to the lower limit of the EKMA model. The average difference between the EKMA model and the linear rollback model was as high as +30%. Hence, we estimate an upper error limit for the rollback model to be 100%.

In principle, statistical models should be able to do somewhat better than linear rollback simply because they use some additional parameters. This of course will not be true for all statistical models. A summary of the uncertainties associated with these four types of models is shown in Fig. 4-14. It should be noted that Fig. 4-14 does not include the impacts of potentially large errors in emissions inventory, for example, +60%.

In actual practice these models may agree better than expected on the basis of the information shown in Fig. 4-14. For example, a comparison of the results using EKMA by the South Coast Air Quality Management District (SCAQMD), the Air Resources Board of California (ARB), the California Institute of TEchnology (CIT) trajectory model and linear rollback shown in Table 4-2 shows an amazing degree of agreement. Similar results are found for July 13, 1978, comparing the ARB EKMA, the CIT trajectory, and linear rollback.

Now that three air quality models and the parameters that influence them have been examined, it is important to examine the ability of ambient air monitoring instruments to measure the pollutants in the air. The precision of these instruments is presented in the next section. In addition, the influence of meteorology on air quality trends is discussed.

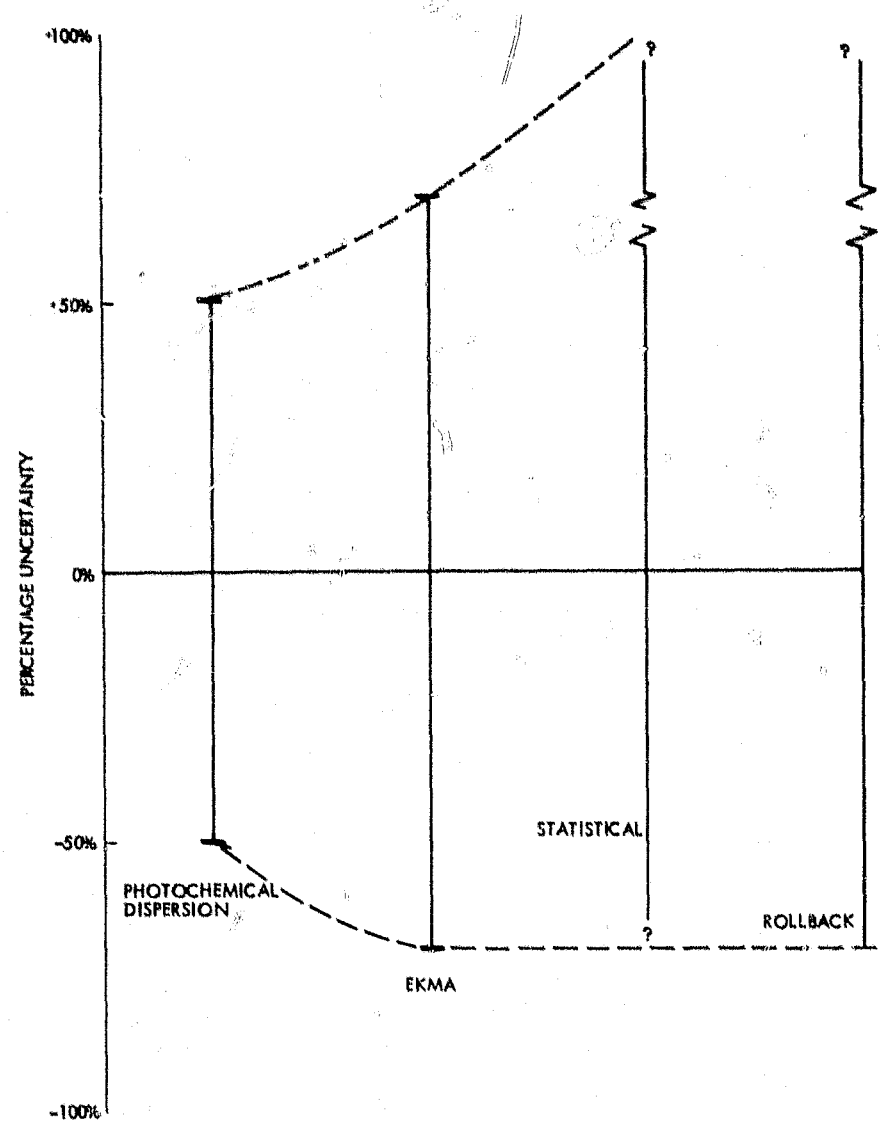


Fig. 4-14, Percentage Uncertainty in Air Quality Modeling Approaches

Table 4-2. Summary of RHC Emission Reductions Required With 40% NO_x Control to Meet NAAQS For Ozone (.12PPM) (DoIA to Upland)

June 27, 1974	
SCAQMD EKMA	80%
ARB EKMA	79
CIT trajectory	77
Linear rollback	83
July 13, 1978	
ARB EKMA	78
CIT trajectory	77
Linear rollback	78

REFERENCES

4-1 Meteorology and Atomic Energy, 1968, David H. Slade, editor U.S. Atomic Energy Commission, Division of Technical Information, July 1968, pg. 1.

4-2 Workbook of Atmospheric Dispersion Estimates, D. Bruce Turner Environmental Protection Agency, Research Triangle Park, NC, pg. 6 (revised 1970).

4-3 Air Quality Management Plan, SCAQMD and SCAG, El Monte, Calif., p. IV-1, Jan. 1979.

4-4 Air Quality Management Plan, SCAQMD and SCAG, El Monte, Calif., p. IV-20, Jan. 1979.

4-5 Souten, D.R., et al., National Commission on Air Quality Los Angeles Regional Study--SIP Process Review, Vol. 1: Review of the SIP Process and Framework and Review and Analysis of South Coast Air Quality Management District Air Quality Management Plan Modeling Methods. SAI, Feb. 1980.

4-6 Goodin, W.P., McRae, G.J., and Seinfeld, J.H., "An Objective Analysis Technique for Constructing Three-Dimensional, Urban-Scale Wind Fields," Environmental Quality Lab, California Institute of Technology, Pasadena, California.

4-7 Uses, Limitations and Technical Basis of Procedures for Quantifying Relationships Between Photochemical Oxidants and Precursors, EPA Office of Air Quality Planning & Standards, Research Triangle Park, NC, EPA-450/2-77-021a, Nov. 1977.

4-8 Falls, Andrew H., McRae, Gregory J., and Seinfeld, John H., "Sensitivity and Uncertainty of Reaction Mechanisms for Photochemical Air Pollution," Department of Chemical Engineering, California Institute of Technology, Pasadena, California 91125 (to appear in International Journal of Kinetics, November 1980 issue).

4-9 Turner, D.B., "Atmospheric Dispersion Model--A Critical Review," Journal of the Air Pollution Control Association, Vol. 29, No. 5, pp. 502-519, May 1979.

4-10 "Atmospheric Dispersion Modeling--A Critical Review," Discussion Papers, Journal of the Air Pollution Control Association, Vol. 29, No. 9, pp. 927-941, Sept. 1979.

4-11 Whitten, G., and Hugo, H., Users Manual for Kinetic Model and Ozone Isopleth Plotting Package, Environmental Sciences Research Laboratory, ORD, EPA, Research Triangle Park, NC EPA-600/8-78-014a, July 1978.

4-12 "Uses, Limitations and Technical Basis of Procedures for Quantifying Relationships Bewteen Photochemical Oxidants and Precursors," EPA-450/2-77-021a, Nov. 1977, p. 34.

4-13 Private communication, A. Ranzieri to T. English. April 11, 1980.

4-14 "Users Manual for Kinetics Model and Ozone Isopleth Plotting Package," EPA-600/8-78-014a, EPA at Research Triangle Park, NC, July 1978.

4-15 McRae, G.J., Goodin, W.R., and Seinfeld, J.H., "Development of a Second-Generation Airshed Model for the Photochemical Air Pollution," Fourth Symposium on Turbulence, Diffusion and Air Pollution, Jan. 15-18, 1979, Reno, Nevada. Published by the American Meteorological Society, Boston, MA.

4-16 McRae, G.J., "Mathematical Modeling of Photochemical Air Pollution," Environmental Quality Laboratory, AIAA paper 314-40, California Institute of Technology, November 1979.

4-17 Reynolds, S.D., et al., Photochemical Modeling of Transportation Control Strategies--Vol. I. Model Development, Performance Evaluation, and Strategy Assessment. Draft Final Report for Federal Highway Administration, March, 1979.

4-18 Martinez, J.R., and Nordsieck, R.A., Evaluation of a Diffusion Model for Photochemical Smog Simulation--Final Report, A.Q. Eschenroeder, EPA-R4-73-012a, October, 1972.

4-19 Mathematical Modeling of Urban Pollution--General Theory, Environmental Science and Technology, 1973, Vol. 7, pp. 253-261.

4-20 Goodin, W.R., McRae, G.J., and Seinfeld, J.H., "Validity and Accuracy of Atmospheric Air Quality Models," Proceedings Third Symposium on Atmospheric Turbulence, Diffusion, and Air Quality, American Meteorological Society, 1976, p. 366.

4-21 Goodin, W.R., McRae, G.J., and Seinfeld, J.H., "An Objective Analysis Technique for Constructing Three-Dimensional Urban Scale Wind Fields," Environmental Quality Laboratory, California Institute of Technology, Pasadena, California.

4-22 Seinfeld, J.H., "Current Air Quality Simulation Model Utility," Dept. of Chemical Engineering, California Institute of TEchnology, Pasadena, California, 1977.

4-23 "Technical Support Document--Phoenix Air Quality Maintenance Plan for Carbon Monoxide and Photochemical Oxidants," Arizona Dept of Transportation, Transportation Planning Div., March 1978, pg. 58.

4-24 Falls, Andrew H., McRae, Gregory J., and Seinfeld, John H., "Sensitivity and Uncertainty of REaction Mechanisms for Photochemical Air Pollution," Dept of Chemical Engineering, California Institute of Technology, Pasadena, California.

4-25 Hatmaker, M.L., "Sensitivity and Sensibility in Air Quality Models," Specialty Conference: Transportation and the 1977 Clean Air Act Amendments, American Society of Civil Engineers, Nov. 12-14, 1979, San Francisco.

4-26 "The Quality of Data and the Choice and Design of Predicted Models," Highway Research Board, Special Report, 97 National Academy of Sciences, Washington, D.C., pp. 178-192, 1968.

4-27 Meteorology and Atomic Energy, 1968, United States Atomic Energy Commission, Division of Technical Information, July 1968, p. 39.

SECTION 5

MEASUREMENT ACCURACY AND TRENDS

In this section both air quality measurements and trends are examined together. This is important to do because the air quality measurements are used as the basis for any kind of trend estimates. Air quality measurement is still a relatively young field. Hence, there are changes being made in the instrumentation. These changes can have both quantitative and qualitative impacts on measurement trends. A recent discussion of air quality measurements appears in Ref. 5-1. This critical review provides much of the background issues in air quality measurements. Since this material is covered so well there is no point in going over these issues in this report. In this section we examine the error limits imposed on several air quality measurements by agencies such as the Environmental Protection Agency and the South Coast Air Quality Management District. These examples will be best case examples since both organizations represent very advanced organizations from the viewpoint of measuring air quality. We shall examine measurements of the oxides of nitrogen, non-methane hydrocarbons, and the implication of these measurements to the ratio of non-methane hydrocarbons to NO_X. We shall also examine the errors associated with ozone measurements.

5.1 NITROGEN DIOXIDE AND NITRIC OXIDE

In order to determine how well a given air quality monitoring instrument such as a nitrogen dioxide instrument is working, standards are brought into the air quality monitoring station and the measurements on a given instrument are compared with the output of the standard. A certain set of rules is established to determine whether or not the instrument is within allowable error limits. As an example of these rules, consider the rules given by the South Coast Air Quality Management District (Ref. 5-1).

The South Coast Air Quality Management District specifies that a set of calibrated values of NO_2 be checked against those measured with a monitoring instrument in the "as found" condition. This check is done by fitting the best straight line to a graph of output of the instruments vs the calibration values of NO_2. This fit is performed by means of a linear regression approach. By means of linear regression the best values of slope and zero offset are determined. The air quality management district considers the data that the instrument has measured since its last check to be good if (1) the offset is less than or equal to $\pm.03$ ppm of NO_2, and (2) the slope deviates from unity by no more than $\pm 10\%$. An example of applying these two rules is shown in Fig. 5-1. The solid lines are for the zero offset case. For example, if the calibrated value of NO_2 was .05, the actual value could range between .45 and .55 and be within the acceptable limits. The +.03 offset case is shown as short dashed lines in Fig. 5-1. In this case

if the calibration value of NO_2 is .5, the acceptable limits would be between .48 and .58. A similar approach applies for a negative .03 offset producing the long dashed lines. Hence, if the regression lines associated with a given instrument fell within either of the three bands, they would be acceptable. Thus, a given instrument may have readings that fall in between the outer lines indicated by the calibration error bar at about .4 ppm from the calibration system. In actual practice, points may lie outside of these lines as long as the overall slope and intercept as determined by linear regression fall within the specifications of ±10% on slope and ±.03 ppm on offset.

Fig. 5-1 gives us a good indication of the in-practice acceptable errors in NO_2 measurements that are made by a well-qualified organization. In this analysis the assumption will be made that these outer limits represent approximately the +2 σ limits for the measurements, that is, approximately 95% of the data falls within the calibration error bars shown in Fig. 5-1.

It should be noted that the above discussion does not make any comments about absolute accuracy of past NO_2 measurements. In making measurements of NO_2 the calibrations have shifted significantly over the years. EPA had a very serious problem with the Jacobs-Hochheiser technique (Ref. 5-2) which was ultimately rejected as a useful technique for measuring NO_2 after many years of measurements (Ref. 5-3). Other approaches such as the Saltzman approach and the gas phase Chemiluminescence approach have been used instead. The Chemiluminescence approach to measurements of oxides of nitrogen has been designated by EPA as the reference measurement principle (Ref. 5-4). The calibration techniques being used in advanced monitoring networks refer the calibrations back to National Bureau of Standards. A discussion of the changes in bias caused by shifting calibration standards is presented in Refs. 5-5 and 5-6. In this discussion Crowe points out that in order to use 1975-79 California NO_x data it must be multiplied by .83. This is caused by a systematic bias in the calibration technique.

If the calibration error limits are taken from Fig. 5-1 and divided by the calibration concentrations of NO_2, a percentage error that is considered acceptable in practice as a function of the concentration of NO_2 is obtained. A plot of this acceptable percentage error vs the concentration of oxides of nitrogen is shown in Fig. 5-2. It may be seen that the error approaches ±15% for relatively high values of NO_x, i.e., .7 ppm. For NO_x values of the order of 0.2 ppm the error is approximately ±25%. For smaller concentrations the error becomes increasingly large, as is shown in Fig. 5-2. This curve will be used later in conjunction with a similar curve for the non-methane hydrocarbons, in order to assess the uncertainty in the ratio of non-methane hydrocarbons to NO_x.

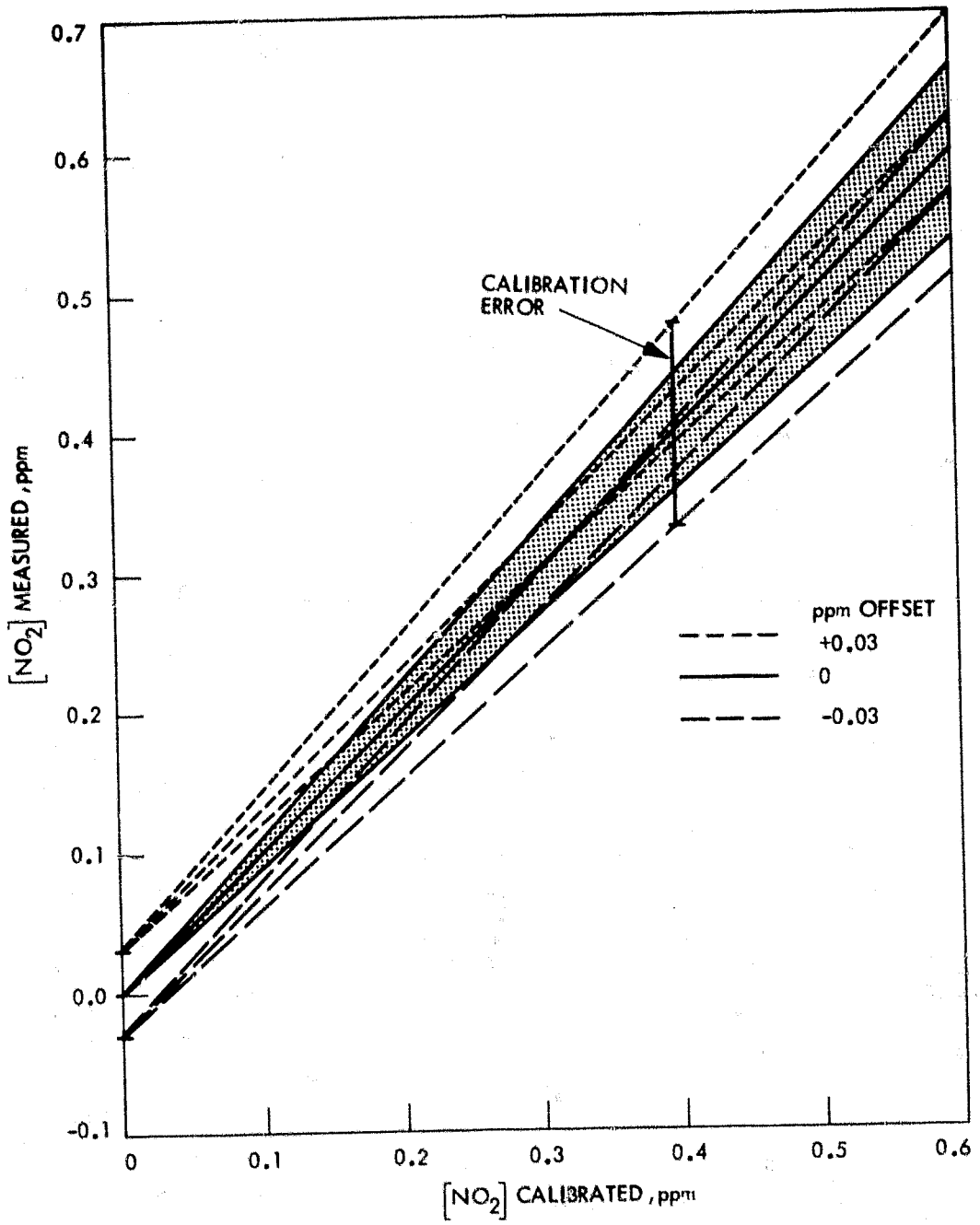


Fig. 5-1. Comparison of Measured versus Calibrated NO_2 Concentrations Considering Calibration Error

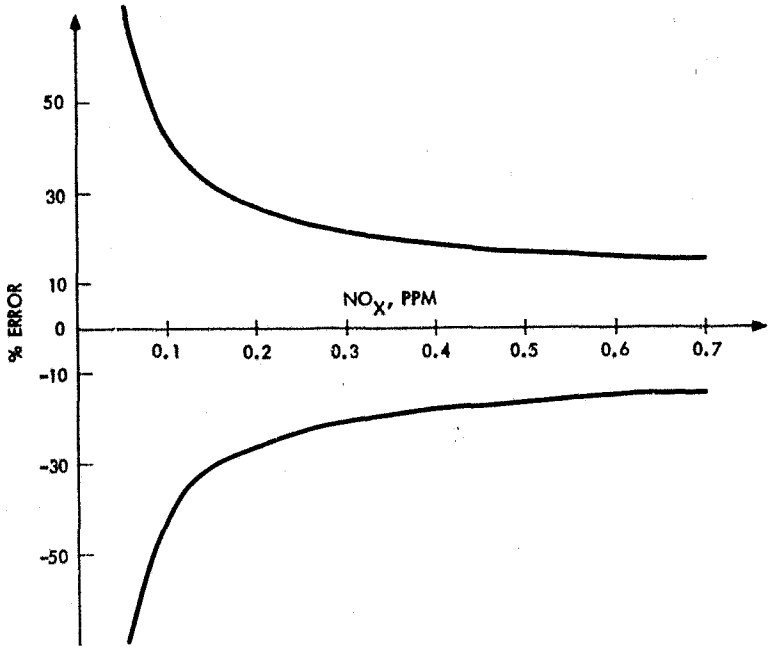


Fig. 5-2. Percent Error versus NO_X Concentration

5.2 NON-METHANE HYDROCARBONS

Similar calibration procedures to those described above for the measurements of oxides of nitrogen can be used for non-methane hydrocarbon measurements. The non-methane hydrocarbon instrument is considerably poorer than the nitrogen oxides measurement described earlier because the technique of measuring non-methane hydrocarbons requires the treating of many, many individual hydrocarbons as though they were quite similar. A discussion of this can be found in the EPA Research Triangle Institute report (Ref. 5-7). The Environmental Protection Agency, in the users' manual for EKMA (Ref. 5-8) specifies the error associated with the non-methane hydrocarbon measurements as being +5% to 10% of the 10 ppm full scale. If the +5% data is plotted in a manner similar to that shown previously for the oxides of nitrogen in Fig. 5-2, a curve shown in Fig. 5-3 is obtained. The solid curves show percentage error as a function of non-methane hydrocarbon concentration in parts per million of carbon.

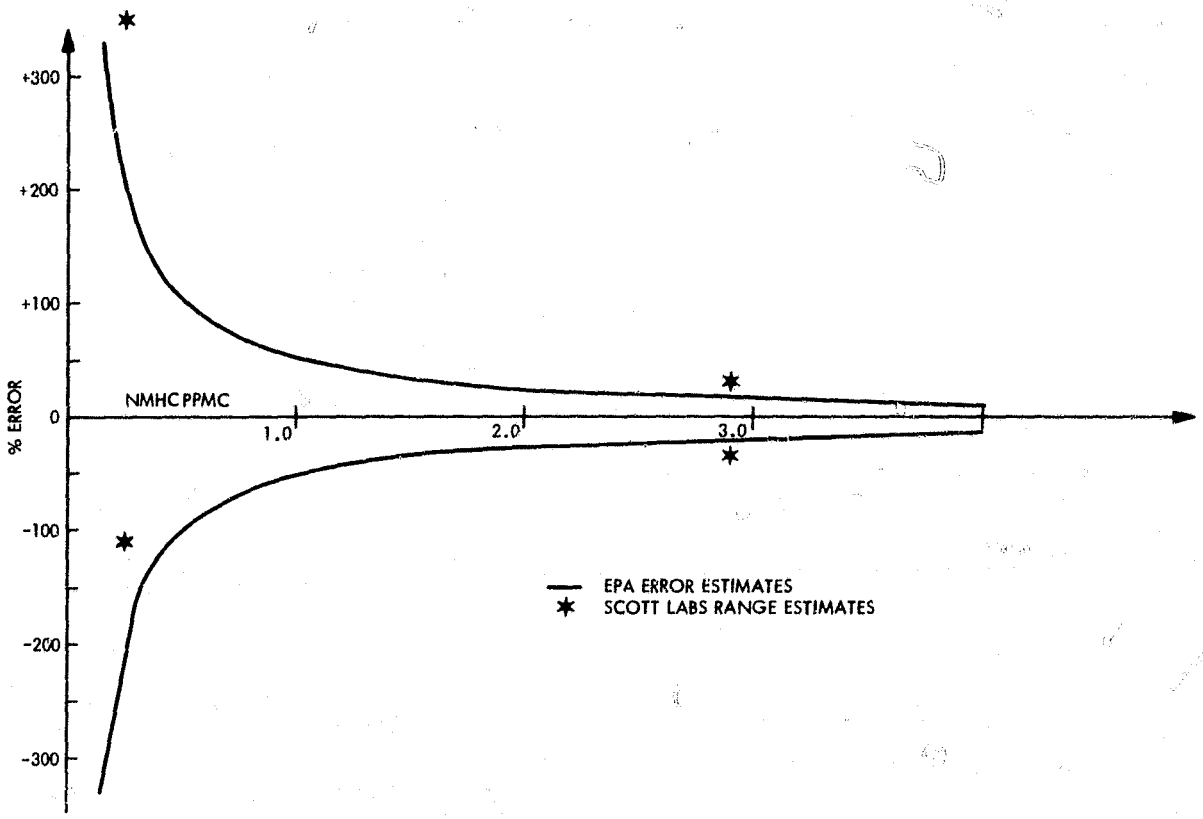


Fig. 5-3. Percent Error versus NMHC Concentration

In order to check these curves, estimates that were made on 13 instruments by Scott Laboratories were used. The range of the estimates is shown in asterisks for 2 calibration values of non-methane hydrocarbon, 2.90 ppmC and .25 ppmC. It may be seen that these range estimates fall quite close to the estimate based on the error estimate of ±5% of full scale as specified in the EPA document. This is not too surprising since the range can be used as an estimate of the standard deviation. For example, see Dixon and Maxxey (Ref. 5-9). For a sample size of 13 measurements, the standard deviation would be about .30 times the range. In keeping with the concept of 90% of the data falling within an error bar, we would use twice the standard deviation, that is, .60 times the range.

In practice, a value of non-methane hydrocarbons of 1 ppmC, which is not unusually low in the Los Angeles area, has an error range from Fig. 5-3 of ±60%. In order for the South Coast Air Basin to reach the ozone standard, it is expected that the non-methane hydrocarbons need to be reduced to about .5 ppmC. At this concentration the error is approximately ±130%. Hence, it can be seen that the errors in non-methane hydrocarbons are quite significant at concentrations of interest to air quality managers.

There is a significant need to improve the abilities of instruments to measure non-methane hydrocarbons. The present generation of instruments appears to be considered dangerous and also highly inaccurate at the concentration levels of interest. An indication of the frustration of air quality investigators with the performance of non-methane is given in Ref. 5-10.

> In summary, it is my conviction that whereas FID's probably respond reasonably uniformly to paraffins they vary significantly in their response to olefins, acetylenes, aromatics and other organics such as the alcohols, ketones and aldehydes. Present instrumentation does not allow legitimate comparison of total hydrocarbon less methane data collected by gas chromatographs designed to produce those data, such as the Beckman 680C, Bendix 8200, and Byron 200 series or the MSA 11-2 and the Bendix 8201.
>
> To add to the complexity of the above stated problem is the recognized fact that the photochemical reactivity of different hydrocarbons varies tremendously.
>
> It is my belief EPA must face up to the stated problem. There are solutions. But irrespective of the solution, I can not foresee any being free of arbitration. Until a solution is agreed upon, and backed by sound research, perpetuation of any numerical concentration limits for hydrocarbons corrected for methane now appears inane. Within this past week I have seen one analyzer produce readings that averaged for one hour a value of 0.22 ppm carbon and another analyzer producing a value in excess of an average of 1 ppm for the same time period for the same air sample.

These errors associated with NMHC measurement instruments do have qualitative impacts on decision making in air quality management. Hence considerable effort should be given by the Environmental Protection Agency to improve the ability of these instruments to measure non-methane hydrocarbons more precisely.

5.3 NMHC/NO_x RATIO

We have examined the errors associated with routine operation of both oxides of nitrogen instruments and non-methane hydrocarbon instruments. In the EKMA model one of the fundamental tools used is the ratio of the non-methane hydrocarbon to the oxides of nitrogen NMHC/NO_x. Since this ratio is so critical to the EKMA analysis we shall examine it and determine the errors in the ratio as determined by the errors in both instruments for measuring NO_x and the instrument for measuring the non-methane hydrocarbons. In Eq. (5-1) we let Ψ equal the ratio of non-methane hydrocarbons to oxides of nitrogen.

$$\Psi = \frac{NMHC}{NO_x} \qquad (5\text{-}1)$$

If we perform a standard error analysis on Ψ in terms of the errors in other components we find that

$$\Delta\Psi = \frac{\partial\Psi}{\partial NMHC}\,\Delta\,(NMHC) + \frac{\partial\Psi}{\partial NO_x}\,\Delta\,(NO_x) \qquad (5\text{-}2)$$

If Eq. (5-2) is divided by Eq. (5-1) we arrive at the relative error shown in Eq. (5-3).

$$\frac{\Delta\Psi}{\Psi} = \frac{\Delta(NMHC)}{NMHC} + \frac{\Delta(NO_x)}{NO_x} \qquad (5\text{-}3)$$

If the assumption is made that the errors in the non-methane hydrocarbon instrument are independent of those errors in the oxides of nitrogen instrument, which appears to be a reasonable assumption, we then have the conditions for orthogonality. Applying the orthogonality concept we arrive at Eq. (5-4), which gives the relative percentage error in Ψ in terms of the errors in both instruments.

$$\left(\frac{\Delta\Psi}{\Psi}\right)\% = \sqrt{\left[\frac{\Delta(NMHC)}{NMHC}\,\%\right]^2 + \left[\frac{\Delta(NO_x)}{NO_x}\right]^2} \qquad (5\text{-}4)$$

Thus we have expressed the percentage error in the ratio in terms of the percentage errors in both the separate instruments.

This information is shown in a graphical manner in Fig. 5-4. In this figure we have arbitrarily picked two typical ratios of NMHC divided by NO_x. Values of 5 to 1 and 10 to 1 have been shown. This curve indicates the percentage error in the ratio as a function of the nominal value of non-methane hydrocarbons. For example, at a NMHC value of 1.0 ppmC ranges of +55 and +65% are obtained, respectively, for the 10 to 1 ratio and the 5 to 1 ratio. Error increases significantly as the nominal value of non-methane hydrocarbons decreases to 0.5 ppmC to $\pm$108 and $\pm$125% for ratios of 10 to 1 and 5 to 1, respectively. A negative error of -125% would indicate that the instrument would actually read a negative number. This of course would have no physical significance. However, this type of behavior is encountered frequently in practice, since the value of non-methane hydrocarbons is determined by subtraction of the methane hydrocarbons from the total value of hydrocarbons. In the subtraction, one large number is subtracted from a second large number. Both of these numbers have relatively large errors. Hence it is not uncommon to obtain NMHC instrument readings which are negative.

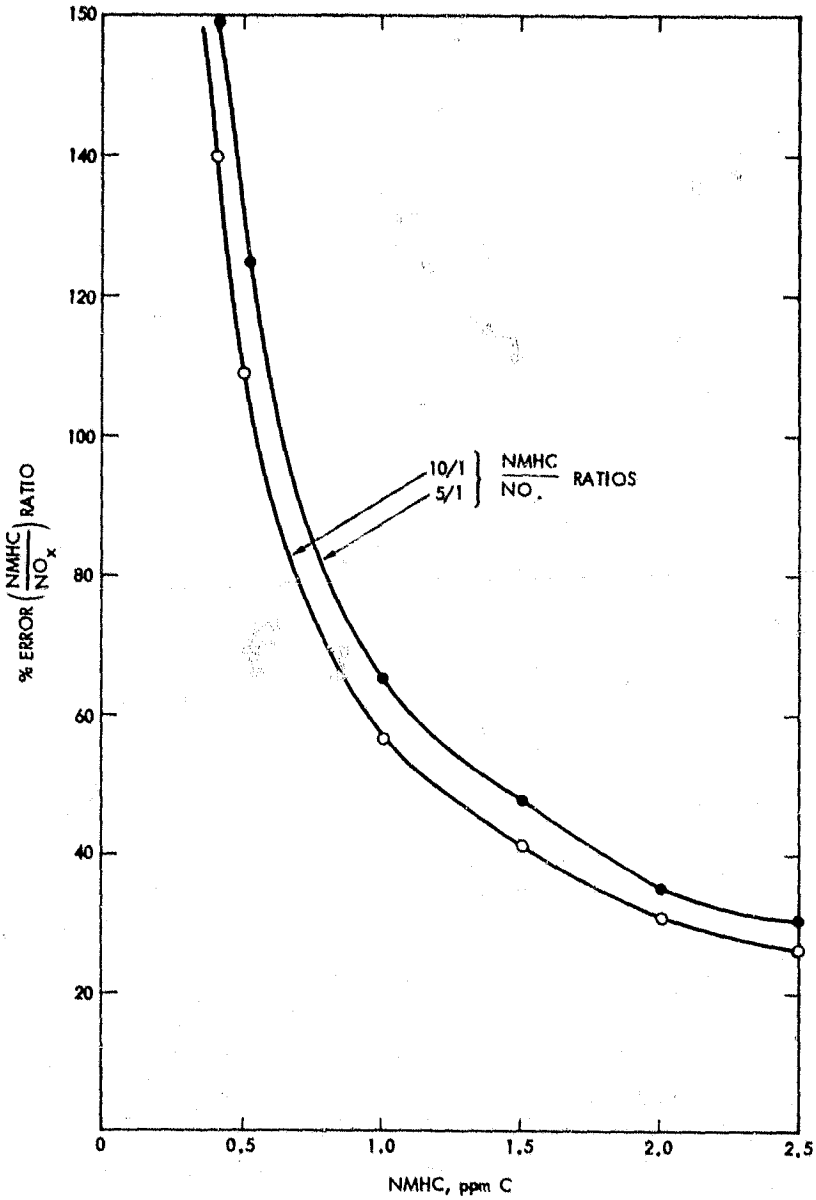


Fig. 5-4. Percent Error in the $NMHC/NO_x$ Ratio versus NMHC Concentration

If the information in Fig. 5-4 is applied to an EKMA type diagram, the results shown in Fig. 5-5 are obtained. On this figure a nominal NMHC/NO_x ratio of 10 to 1 is shown as the center of the diagram. If we examine the .10 ppmC NMHC nominal value, we find that on the basis of Fig. 5-4 there is a ±66% error in the NMHC/NO_x ratio. Hence, the ratio could vary between 16.6 and 3.4 for this nominal value of 10. These data points are plotted on Fig. 5.5. Similar points were plotted for other nominal values of non-methane hydrocarbons. This approach resulted in an NMHC/NO_x error envelope given by the lower and upper bounds shown in Fig. 5-5. As can be seen from this figure, as the nominal values of non-methane hydrocarbon approach those expected in the ambient environment, the error in the ratio becomes highly significant. This type of highly significant error introduces great uncertainty into the air quality management aspects of using the EKMA analysis.

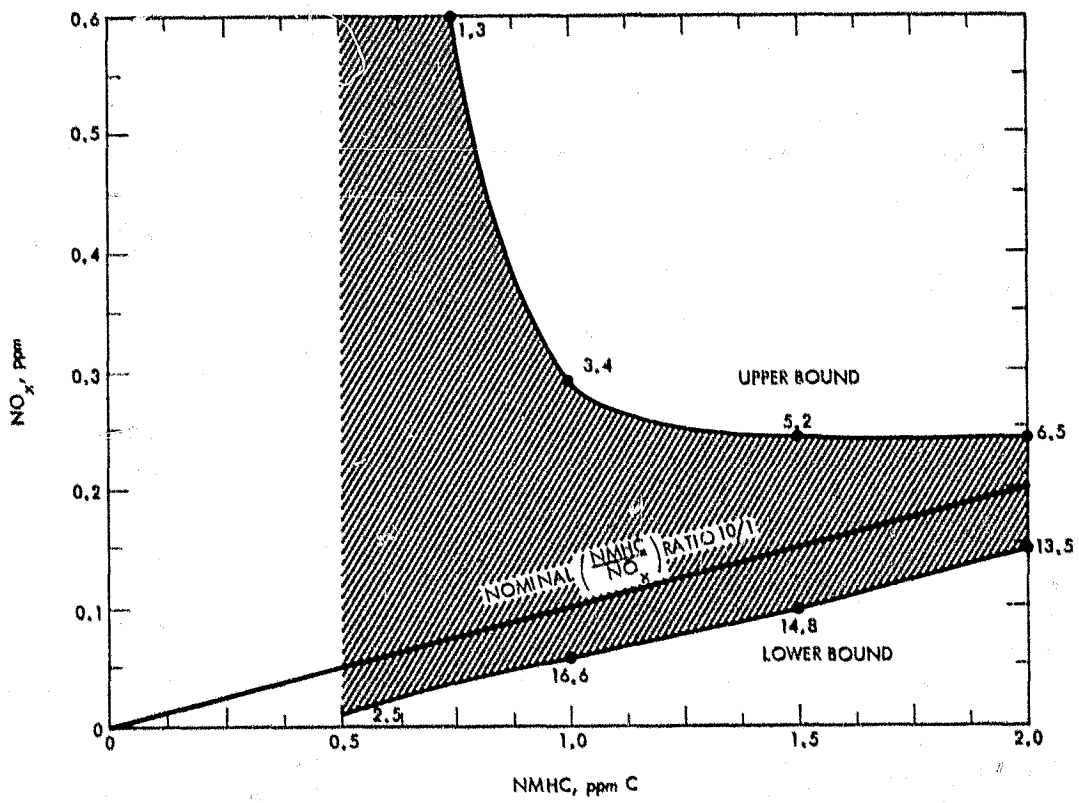


Fig. 5-5. Upper and Lower Bounds on the NMHC/NO_x Ratio for a Nominal Ratio of 10

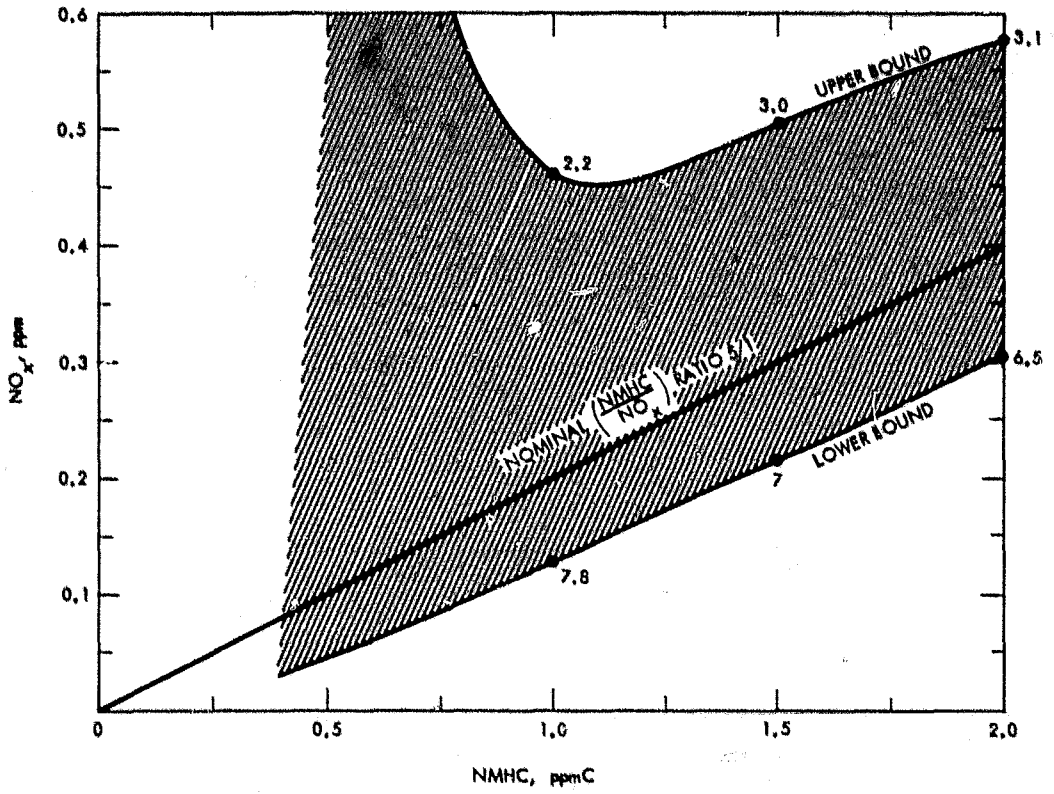


Fig. 5-6. Upper and Lower Bounds on the NMHC/NO_x Ratio for a Nominal Ratio of 5

A similar curve is shown in Fig. 5-6 for a nominal NMHC/NO_x ratio of 5 to 1. Once again, if we take 1.0 ppmC NMHC as a reference point, we obtain a range of ±56%. Since the nominal ratio is 5 to 1, the value of NMHC/NO_x ranges between the values of 2.2 and 7.8, which are shown as dots on Fig. 5-6. By repeating this process, lower and upper bounds are obtained to describe the limits associated with the nominal ratio of 5-1. Once again as values of NMHC approach these expected in ambient air in many urban areas, one finds very large errors in the ratio of NMHC to NO_x.

5.4 OZONE MEASUREMENTS

A discussion of the ozone reference measurement methods is presented in Ref. 5-11. Ozone has had considerable changes in the way in which it was measured. Early measurements were made with phenothalene. This was replaced by measurements using potassium iodide. Some investigators would buffer the potassium iodide, other investigators, such as the Los Angeles Air Pollution District, would use unbuffered postassium iodide. This resulted in a considerable difference in readings between the California Air Resources Board and the Los Angeles Air Pollution Control District at nearby stations. A study was conducted several years ago which basically led to the elimination of this problem by means of shifting the technique for measuring ozone to a

photometric technique. Presently the photometric technique is used in many measuring areas. Results of the O_3 instrument errors that are considered acceptable in practice using this photometric technique have been supplied by John Higachi of the South Coast Air Quality Management District (Ref. 5-1). In making checks of the instruments used in the air quality monitoring network by the South Coast Air Quality Management District during 1979, 53 calibration checks were made. The results of these checks indicated that the mean deviation of the instruments from the standards (which are referred back to the National Bureau of Standards) was .2%. In a measurement of the mean, positive errors tend to cancel negative errors. Perhaps a more meaningful measurement is the standard deviation associated with a comparison of the actual monitoring instruments and the reference standards. In this case a standard deviation of 6% was obtained.

Another approach used for ozone instruments is to make a plot of the concentration measured with the monitoring instrument vs the concentration obtained through the reference standard. If these values agreed perfectly, the result of this plot would be a 45° line with the slope equal to 1. One indication of the degree of difference between the two would be to measure the slope of such a curve. This is particularly useful in the case of ozone measurements since zero offsets with the ultraviolet photometric instruments are usually quite small. Such an investigation has been conducted by the South Coast Air Quality Management District and the results are presented in Fig. 5-7. This figure shows a plot of the percentage error in the ozone slope as a function of the cumulative percentage of the measurements which have a slope error less than or equal to the values shown. By examining Fig. 5-7 one sees that 60% of the measurements have a slope error which is less than ±5%, and approximately 93% of the measurements

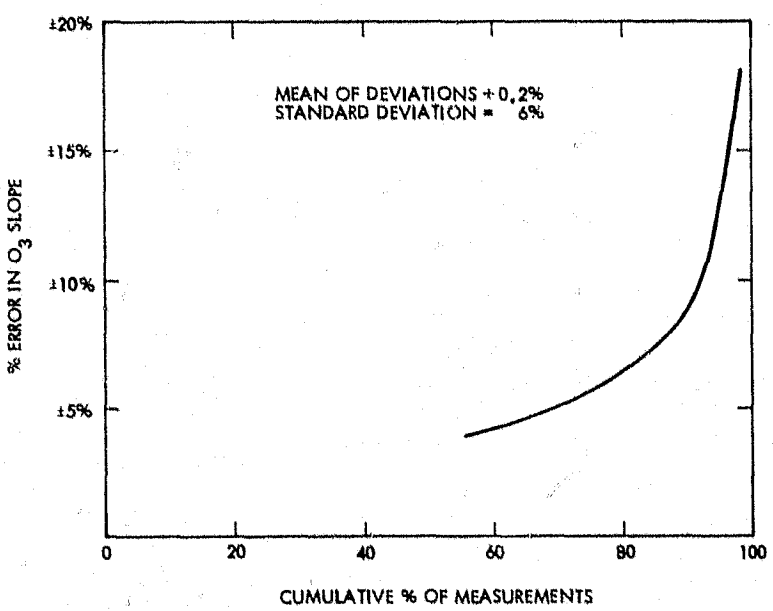


Fig. 5-7. Cumulative Percent of Meausrements

have an error in slope which is less than ±10%. This indicates that the relative errors in ozone measurements are quite small in comparison with those for the oxides of nitrogen, and in particular for those in non-methane hydrocarbons. Hence, it does not appear that the measurement of ozone causes any qualitative impact on air quality management.

5.5 AIR QUALITY TRENDS

Whether the air quality in a given region is improving, staying the same, or degrading is typically determined by the use of air quality trends. These trends are determined by taking measurements and plotting these measurements for various years to determine whether changes are taking place. The many factors that affect air quality concentrations have been discussed previously, e.g., see Fig. 4-8 in Section 4. The air quality is determined by the meteorology, the emissions, and the chemistry that occurs. If one basically ignores these three major inputs and simply plots a curve of air pollutant concentration vs year, one may arrive at surprising results. For example, in the data shown in Fig. 5-8 it was concluded in 1976 that indeed the air quality in the South Coast Air Basin was improving dramatically. This conclusion was based on a simple extrapolation of the end points of the graph. Unfortunately, it turned out that in the subsequent years the air quality worsened such that the conclusions drawn on the basis of Fig. 5-8 were inappropriate.

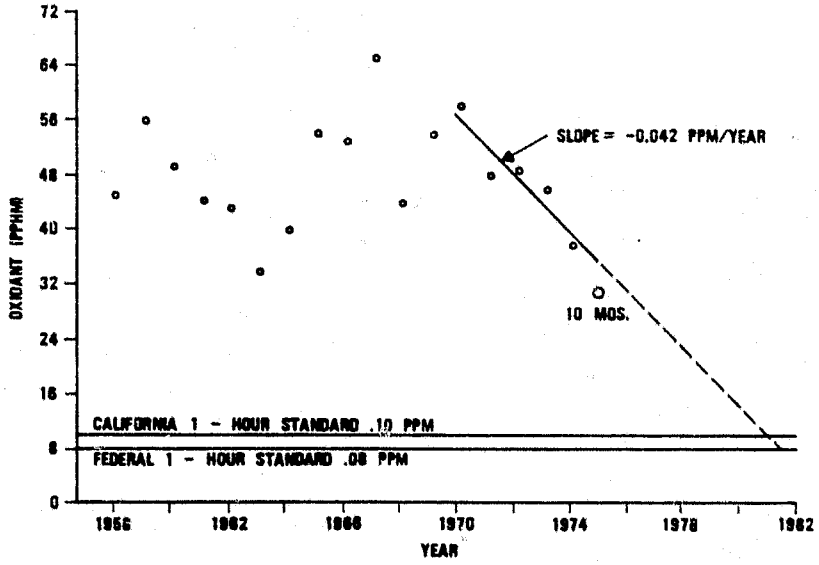


Fig. 5-8. Oxidant Trend Annual 1 - Hour Maximum, East San Gabriel Valley

NOTE: This figure is copied from a 1975 publication describing oxidant trends in the South Coast Air Basin. As shown, the slope of the "projected" line is very sensitive to the number of years used in its determination. (SCAPCD 1975)

At a recent conference on long range trends at Caltech (Ref. 5-12), it was concluded that very little, if any, improvement has occurred in air quality in the South Coast Air Basin over the past 15 years. However, this conclusion was not based simply on fitting a curve to the data. For example, an analysis was performed by G. Cass et al. (Ref. 5-13) to adjust the air quality data for meteorological factors such as those shown in Fig. 4-8 in Section 4. Cass's analysis included adjusting the data for inversion height, radiation intensity, wind speed and temperature. No attempt was made to include the effects of cloud cover or turbulence directly. This analysis indicated that 70% of the variability in data could be accounted for on the basis of the above meteorological factors. The equation that was used to adjust for the variability is shown in Eq. (5-5).

$$O_3 \propto \frac{K_1^{0.74} \, F(T)^{1.58}}{\bar{U}^{0.46} \, Z^{0.40}} \tag{5-5}$$

where K_1 is the sunlight dependent rate constant described in Section 4.2, $\bar{U}$ is the average wind speed, Z is the inversion height, and F(T) is a temperature-dependent chemical reactivity. The exponents used in Eq. (5-5) were chosen to statistically maximize the correlation coefficient. Taking the total derivative of the ozone in Eq. (5-5), one finds that the percentage change in ozone concentration can be expressed as Eq. (5-6).

$$\frac{\Delta O_3}{O_3} = \frac{\Delta K_1}{O_3}\frac{\partial O_3}{\partial K_1} + \frac{\Delta F}{O_3}\frac{\partial O_3}{\partial F} + \frac{\Delta \bar{U}}{O_3}\frac{\partial O_3}{\partial \bar{U}} + \frac{\Delta Z}{O_3}\frac{\partial O_3}{\partial Z} \tag{5-6}$$

If the assumption is made that the errors in the rate constant K_1, the function of temperature F(T), the wind speed $\bar{U}$, and the inversion height Z are independent, then orthogonality applies to Eq. (6). Hence the percentage error in ozone due to changes in the above meteorological factors can be expressed by Eq. (5-7).

$$\frac{\Delta O_3}{O_3} = \left[\left(.74\,\frac{\Delta K_1}{K_1}\right)^2 + \left(.46\,\frac{\Delta \bar{U}}{\bar{U}}\right)^2 + \left(1.58\,\frac{\Delta F}{F}\right)^2 + \left(.40\,\frac{\Delta Z}{Z}\right)\right]^{1/2} \qquad (5\text{-}7)$$

In Ref. 4-22, J. Seinfeld makes estimates of these errors as follows. The error in the rate constant K_1 is $\pm 20\%$. The error in the wind speed is $\pm 20\%$. The error in the inversion height is $\pm 25\%$. Seinfeld makes no estimates of the error in the function of temperature. For the sake of this analysis let's assume that the error in the function of temperature is on the order of $\pm 5\%$. If these errors are substituted into Eq. (5-7), one arrives at an overall net error due to errors in these meteorological factors of $\pm 21.6\%$.

Thus, this example shows the intrinsic complexity of adding errors which are independent errors in a complex system. If Eq. (5-5) were changed to some equation which would have a more physical basis, one might use an equation such as Eq. (5-8).

$$O_3 \propto \frac{K_1\ F(T)}{\bar{U}\ \ Z} \qquad (5\text{-}8)$$

In Eq. (5-8) the independence of ozone concentration upon the meteorological factors does not involve powers other than the power of 1. For this case

$$\frac{\Delta O_3}{O_3} = \left[\left(\frac{\Delta K_1}{K_1}\right)^2 + \left(\frac{\Delta \bar{U}}{\bar{U}}\right)^2 + \left(\frac{\Delta F}{F}\right)^2 + \left(\frac{\Delta Z}{Z}\right)^2\right]^{1/2} \qquad (5\text{-}9)$$

Substituting the percentage errors of the meteorological factors that were described previously, this combination of effective uncertainty produces an overall uncertainty of approximately 38.1%. Hence, by using very simple linear terms in Eq. (5-8) as opposed to complicated exponential terms in Eq. (5-5), one finds that the net error in the system is approximately double. It should be noted that in the analysis of Cass et al., Eq. (5-8) accounted for approximately 50% of the variability in the ozone data. In contrast, Eq. (5-5) accounted for 70% of the variability. In either case, using Eq. (5-5) or Eq. (5-8), a highly significant amount of the variability is accounted for by meteorological factors. Hence, these results argue strongly that it is quite useful to adjust the air quality monitoring trend data in order to account for the effects of changes in meteorology from year to year.

This section has examined the air quality management uncertainties due to ambient air pollution monitoring instrumentation. In earlier sections the influences of uncertainties in emissions inventories and modeling were presented. The next section demonstrates the effect of these uncertainties on the abilities of the air quality managers to predict the impact of their strategies for improving air quality.

REFERENCES

5-1 Private Communication, John Higachi to T.D. English, May 21, 1980.

5-2 Perkins, H.C., Air Pollution, McGraw Hill, 1974, pg. 301.

5-3 Hauser, T.R., and Shy, C.M., "Position Paper: NO_x Measurement," Environmental Science Technology, Vol. 6, pp. 890-894, 1972.

5-4 Katz, M., "Advances in the Analysis of Air Contaminants--A Critical Review," Journal Air Pollution Control Association, Vol. 30, No. 5, May 1980.

5-5 "California Air Quality Data, Jan, Feb, March 1977," California Air Resources Board, Vol. IX, No. 1, pg. 2.

5-6 Crowe, D., "Trends of California Urban Air Quality in the 1970's," Conference on Air Quality Trends in the South Coast Air Basin, California Institute of Technology, February 21-22, 1980.

5-7 Harrison, J.W., Timmons, M.L., Benyszyn, R.B. and Decker, C.E., "Evaluation of the EPA Reference Method for Measurement of Non-Methane Hydrocarbons," EPA-600/4-77-033, June 1977.

5-8 "Uses, Limitations and Technical Basis of Procedures for Quantifying Relationships Between Photochemical Oxidants and Precursors," EPA-450/2-77-0219, Nov. 1977, pg. A-9.

5-9 Dixon and Massey, Introduction to Statistical Analysis, Third Edition, 1969. McGraw Hill, Inc., pg. 136.

5-10 Memo from G.C. Ortman to F. Burmann, "Identification of Problem (Hydrocarbons Corrected for Methane Methodology)," April 12, 1974.

5-11 Katz, M., "Advances in the Analysis of Air Contaminants-A Critical Review," Journal Air Pollution Control Association, Vol. 30, No. 5, May 1980, pp. 528-557.

5-12 Conference on Air Quality Trends in the South Coast Air Basin, California Institute of Technology, Pasadena, California. Feb. 21 and 22, 1980.

5-13 Graphic Package for "Meteorological and Chemical Potential for Oxidant Formation," G.R. Cass, H.A. Gray and G.J. McRae at Conference Cited in Ref. A.

SECTION 6

AIR QUALITY MANAGEMENT

The art of air quality management includes many of the factors that have been discussed previously in this report. However, other factors such as political acceptability of control measures may be even more important. Air quality models determine what decreases in NO_x and HC emissions are required in order to meet, for example, the air quality standards for ozone and nitrogen dioxide. Various strategies may be examined to effect these reductions. Each strategy, such as reducing emissions from stationary sources, may be accomplished through a set of specific tactics. Each tactic produces a given emissions reduction. These required emission reductions are examined in terms of their control cost. The control costs are associated with various strategies and various tactics within each strategy. Typically, these costs may be expressed in terms of dollars per ton reduced. Hence, if the $/ton for a given control tactic is considerably more expensive than that of other tactics, it will be rejected. Other control factors may have political limitations on their acceptability. Hence, the politics which govern a given region may pose severe limitations on tactics which are applicable to that region. For example, the development of an acceptable inspection and maintenance plan has not yet succeeded in California.

The Clean Air Act amendments of 1977 require that reasonable further progress be measured in terms of the emissions. This is to be done on an annual basis. Hence, these measurements may give new insights into air quality management and may act as restraints on certain options. And, last but not least, are the various measurements of air quality. After all, air quality is what this is all about. As was noted in Chapter 5, these air quality measurements should be adjusted for meteorology in order that the annual and seasonal trends be more meaningful. This entire process acts to limit the options which might be applicable to solving a given region's air quality problem. A schematic diagram of this process is shown in Fig. 6-1.

It should be noted that this process does not occur simply once. The process should be one in which these air quality management factors are being continuously evaluated. In particular, new evaluations would be appropriate with each reasonable further progress report. Hence, the system really is a moving system in which the air quality problem is continuously changing. The models may change, the political climate may change, and the meteorology may change. Hence, the basis of air quality management is a time-dependent organic system with multiple levels of feedback.

In this section, a summary of the air quality management approaches is made for the three regions under consideration, i.e., the South Coast Air Basin, the Denver Air Basin, and the Phoenix Air Basin. The implications of the earlier sections of this report on these air quality management approaches are mentioned in the appropriate places.

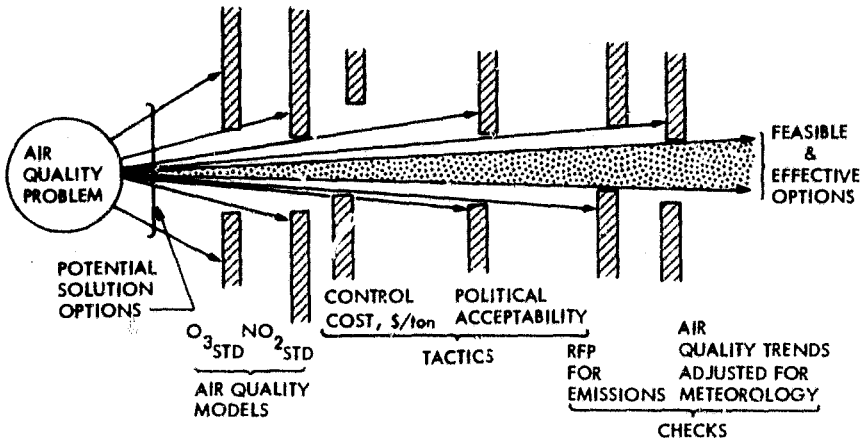


Fig. 6-1. Partial Set of "Decision Making Windows" Which Influence Air Quality Management

6.1 EMISSIONS INVENTORIES

The emissions inventories for the three regions under consideration are described in the references cited previously for these state implementation plans. Each region has estimates of the emissions of the primary pollutants such as SO_2, NO_x, reactive hydrocarbons, etc., which impact the region's air quality. However, none of the three regions studied were able to estimate the degree with which they were certain of the emissions. That is to say, the uncertainties in the emissions inventories are uncertain. Some insight into the uncertainties for stationary sources in the South Coast Air Quality Management District can be obtained through quotation from the SIP process review done for the National Commission on Air Quality (Ref. 6-1).

> Because of the complexity and size of the stationary source emissions inventory data base, it is difficult to quantify problems and errors associated with this inventory. Over 40,000 data entries are included in the stationary source computerized emissions data base, and many of the data entries are very old (i.e., many date back as far as the 1950's). Uncertainties in this data base can be classified into the following three areas of potential error.

(1) Errors in the computer data base.

(2) Omissions of certain equipment categories from the data base.

(3) Gross estimates made because of the lack of data.

One estimate by KVB Corporation of the average degree of uncertainty in the stationary source emissions inventory is as follows:

(1) $\pm$45% for NO_x.

(2) $\pm$60% for total hydrocarbons.

Based on the scatter of the data points shown in Section 2 of this report, and also based on the rather low correlation coefficients that were found in terms of emissions vs vehicle miles traveled (correlation coefficients less than or equal to .3), it is assumed the error in the mobile source inventory is at least this size. In the case of the mobile emission sources, the error is compounded by the fact that the sources move, and their locations can only be estimated by transportation models which also have a certain degree of error.

The authors of this report have not been able to get estimates of the errors associated with the emissions inventories from either the South Coast Air Quality Management District or the Air Pollution Control Division of the State of Colorado or the Arizona Department of Transportation simply because the personnel who would have this information do not know the answers. In order to determine such information, extensive manpower would be required which would require both dollars and time. These resources simply have not been available. Hence, in tracking the reasonable further progress in terms of emissions from year to year, we simply are uncertain about the uncertainty in the emissions. This situation then causes a highly significant uncertainty in terms of analysis of reasonable further progress. In order to minimize these uncertainties, it appears that it may be necessary for the Congress to modify the Clean Air Act in such a way as to require that not only the emissions be estimated for a region on a yearly basis, but that the uncertainties in those emissions also be estimates. If the Congress imposes such a requirement on the Air Quality Management Districts, it would be possible for the Congress to supply the funds which are required to determine these uncertainties. An alternate approach to supplying these funds would be to increase or, in many cases, initiate emission fees on stationary and/or mobile sources of emissions of air pollutants.

In the case of estimates of the reductions of future emissions, there is some information available from the Southern California Association of Governments on their estimates of the uncertainties in various tactics. These emission uncertainties are expressed in terms of both cost of control and also the confidence range of emission reduction estimates. A discussion paper (Ref. 6-2) contains the following statement:

> The quantitative estimates made in these three categories have varying levels of precision and accuracy. Notwithstanding the use of the most up-to-date data base methodologies, all quantitative analysis underlying the AQMP is subject to some uncertainty. To address this uncertainty, confidence ranges surrounding the cost and emission reduction calculations have been estimated based on staff judgment....
>
> The overall reliability of the AQMP (Air Quality Management Plan) inventory has not been assessed quantitatively and there are no available quantitative estimates of the confidence ranges associated with the emission factors for each source type. The AQMP inventory reflects the latest revisions to emission factors approved by EPA and is the basis for estimating the emissions reduction potential of alternative measures (note that in AP-42, each emissions factor is qualitatively ranked on a scale of 1 to 5). Overall, the inventory is generally considered to be the most comprehensive of its kind ever done in the South Coast Air Basin....
>
> Uncertainties in each of these areas have been assessed and the overall confidence ranges of all recommended tactics are presented in Tables 1 and 2. The confidence ranges of 0-20% $\pm$, 20-50%, and greater than 50% themselves reflect the qualitative judgment of technical staff and are not meant to portray an absolute degree of precision.

This type of approach appears to be a step in the right direction. However, it is a step that is clearly needed for the present inventory as well as for projections towards future inventories. Since we have seen that the emissions inventories for both stationary sources and mobile sources are determined largely by modeling considerations, the estimates of any uncertainties in these emissions inventories would also be determined by means of similar models. The data base that has been presented in this report indicates the need to improve our knowledge of the major components of the emission inventory models.

In the Phoenix Air Quality Region, the claim is made that the auto inspection maintenance program has reduced the emissions of carbon monoxide by 22% from 1976 to 1977 (Ref. 6-3). Similar reductions have been reported for New Jersey (Ref. 6-3, p. 3). In the Denver region and the South Coast Air Basin, inspection and maintenance programs are having a difficult time getting started. In the Denver region, the inspection and maintenance program that was put into effect did not require compulsory repair of an auto that was found to be outside of the inspection limits. This lack was ruled to be unacceptable by the Environmental Protection Agency, and hence the economic sanctions mentioned in the introduction of this report were imposed on the Denver area. These sanctions will, hopefully, be lifted as soon as the state of Colorado inserts a requirement for some level of repair upon failure of the emissions test.

In the state of California, the topic of inspection and maintenance has received considerable debate and is the subject of considerable political controversy. Several proposed bills in both the state assembly and the state senate have failed to date. At present, no major construction which requires EPA permits can be initiated within the South Coast Air Basin.

6.2 AIR QUALITY TRENDS

It has been shown earlier in the report that it is very difficult to determine the progress in improving air quality by simply plotting air quality trends. This difficulty is caused by the wide fluctuations in meteorology that may occur from year to year. It has been suggested that this problem can be reduced significantly by adjusting the data for meteorological factors in the manner suggested by Cass et al., as discussed in Section 4 of this report. This kind of adjustment is needed in particular if the rate of improvement of air quality is quite small compared to the size of the problem, which is apparently the case in the South Coast Air Basin. Air quality trends for the South Coast Air Basin have been discussed previously.

6.3 REASONABLE FURTHER PROGRESS TRACKING

As has been discussed previously, the uncertainties associated with the emission inventories are uncertain. Hence, it makes it very difficult to track reasonable further progress by knowing only the estimate of the given emission for a given year. If the agencies were required to estimate the uncertainties associated with the emissions inventory for each pollutant, they would be able to make a statement, with some statistical confidence, of whether they were making progress from year to year. As it is, an emission inventory may increase or decrease from one year to the next and is very difficult to state with any precision whether or not improvement is being made. If the emissions data were plotted in the manner shown in Fig. 6-2, that is with error bars, several interesting improvements could occur. First, the agencies would be able to know what the uncertainty was in their data. Actions could then be taken to reduce this uncertainty on a systematic basis. If the uncertainty became smaller from year to year, this would certainly make it easier to determine statistically whether there were statistically significant trends of improvement.

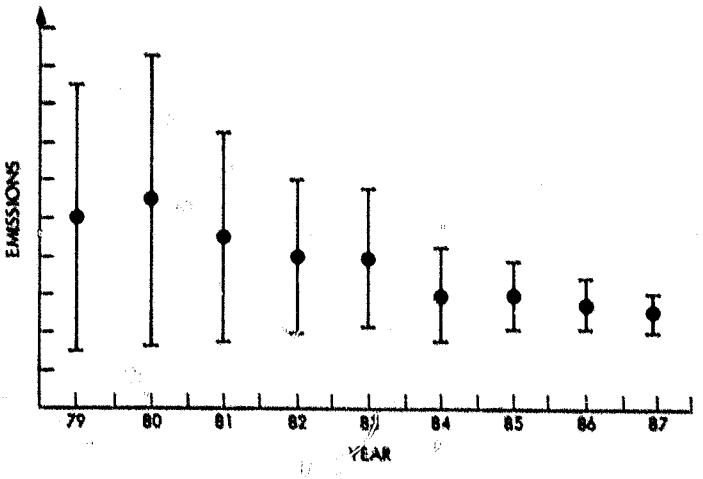


Fig. 6-2. Proposed Modified Reasonable Further Progress Plot Including Estimates of Uncertainties

It would also be useful to both air quality managers and the general public if ambient air quality were added to the requirements for reasonable further progress. Some indication of the error bars associated with the models discussed earlier (for emission inventories which are assumed to be quite accurate, and ignoring meteorological uncertainties) is shown in Fig. 6-3, a plot of the maximum hourly average of ozone vs years into the future for the South Coast Air Basin. The measured ozone value shown for 1979 represents the very bad episodes that occurred in September of 1979. The error bars associated with those measurements are derived from Section 5. In 1982, a hypothetical point is predicted of about .33 ppm maximum hourly average for ozone. This prediction would have different error bars associated with it depending on which model were used. For example, if a photochemical dispersion model were used which had a ±50% uncertainty, the large black error bars would result. However, if the EKMA model were used with its nominal ±70% uncertainty, the thin error bars would result. Hence, in both cases, the spread of the data is quite large.

The situation changes somewhat if we extend the concept of prediction out to 1987 where it is shown that the ozone national ambient air quality is met. In this case, the error bars for both the photochemical dispersion model and the EKMA model are relatively small in absolute value. Hence, one can use these kinds of models to predict gross changes, with confidence, such as those shown from about .45 in 1979 to .12 in 1987. However, for periods in between where the absolute value of the prediction is larger, the errors are correspondingly larger. Hence a combination of emissions reductions measurements and air quality measurements (adjusted for meteorological factors) would be useful to determine if the air quality management techniques that are being used are successful.

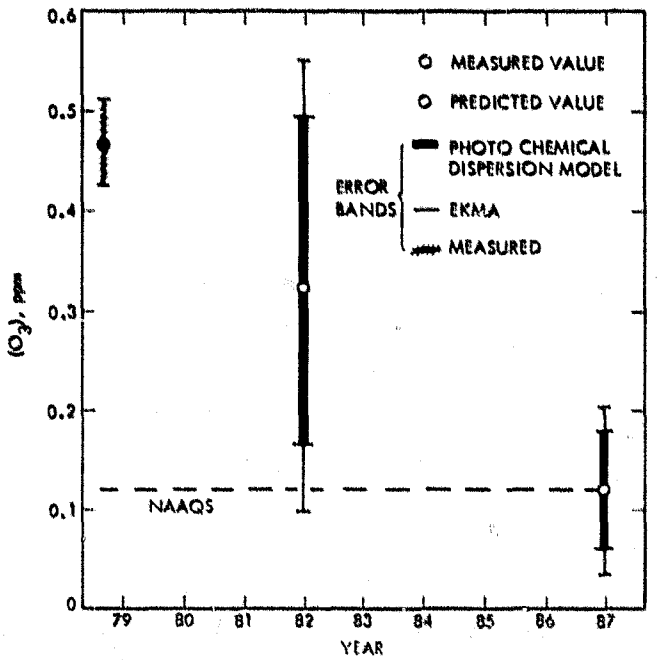


Fig. 6-3. Ozone Error Band Estimates for South Coast Measurements and Models

Several emission-air quality models exist that are capable in principle of predicting the impact of transportation systems on ambient air quality in a given AQMD. In practice, the chief difficulty of predicting impacts lies mainly in the large uncertainties in pollutant emissions inventories, in meteorological data, in rates of chemical reactions of the pollutants in the atmosphere, and in the measurements of ambient air quality itself. In those AQMD's where mobile sources are responsible for most of the pollutants, the uncertainty in emissions of reactive hydrocarbons may be +70% of the mean. Uncertainties in meteorology, including average wind speed, height of the base of the inversion layer, sunlight intensity, and ambient temperature effects, lead to an "error" or uncertainty of +27%. Uncertainties in initial conditions and chemical reaction rates are +50%. As a result, in the early 1980's, the "error bands" in the predicted ambient air quality are probably as large as the mean value (±100%) even for the most complex models. In practice, the impact of a new transportation source may best be treated as a perturbation on the existing air emissions system, unless its emissions constitute a significant percent increase in the emissions of a given district.

The uncertainties associated with present day measurements of both NO_x and non-methane hydrocarbons introduced considerable uncertainties into any model. An example of this uncertainty is applied to EKMA in Fig. 6-4. A similar analysis has been performed by Eschenroeder (Ref. 6-4), in which the percentage errors of NMHC and NO_x measurement have been assumed to be independent of the concentration of the pollutants. The ozone isopleths that are shown in this figure have been adjusted specifically for the South Coast Air Basin. Hence, they appear to be quite different than those seen before in Section 4 on modeling. Two years of interest are shown on the diagram--1974 and 1987. In the air quality management plan submitted by the South Coast Air Quality Management District and the Southern California Association of Governments, 1974 was used as the base year. The non-methane hydrocarbon value for the peak oxidant day was 1.5 ppmC, and the NO_x value was approximately 0.4 ppm. The shaded box indicates the combined effect of error bars in both NO_x and hydrocarbons. It may be seen that the actual value of ozone could lie at values as low as the air quality standard and possibly as high as values of about .7, which is not shown on the graph. Hence, the relative degree of uncertainty in the measurements of both NO_x and NMHC causes considerable uncertainty in terms of the starting point.

As the attempt is made to meet the national ambient standard for ozone in 1987, a nominal point of about .5 ppmC for non-methane hydrocarbons and about .23 for NO_x is reached. Once again, considerable error is shown in the value of non-methane hydrocarbon, which could lie within a two-sigma limit extending from 0 to 1 ppm. This range could cause the value of ozone to vary from about .38 down to background. This instrument uncertainty causes significant uncertainty in terms of the air quality management decisions that must be made. It should be noted that these uncertainty values were taken from the data in Section 5, Measurements and Trends. The error bars shown in Fig. 6-4 do not consider the errors associated with the model itself, that is, ±70%. These shadings assume the model error to be quite small compared to the instrumentation error. Hence, this may be an optimistic presentation.

6.4 AIR QUALITY ISSUES

6.4.1 Management Air Quality Standards

Present ambient air quality standards for oxidant are expressed in terms of a maximum hourly average concentration not to be exceeded more than one day per year.

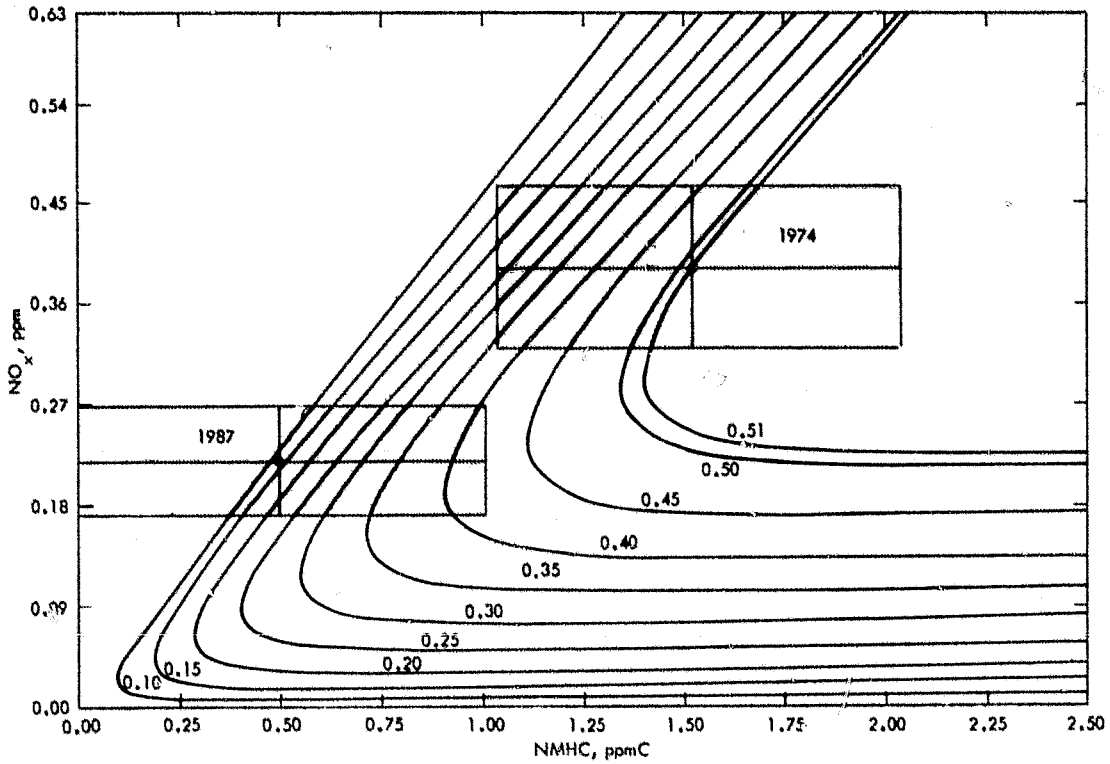


Fig. 6-4. EKMA Plot for South Coast Air Basin Including Measurement Uncertainties.

In the SCAQMD this standard is violated more than 200 days per year, often for many hours per day. In other words, such violations are by no means rare events. The assessment of the emissions air quality models in this report shows that these models are poorly equipped to predict rare events because of the large uncertainties in the data base. A more satisfactory approach is to retain the maximum hourly average concentration (0.12 ppm O_3) but utilize the number of days per year, or hours per year, that this standard is exceeded as a measure of progress toward achieving the air quality standards. For example, at certain stations in the South Coast Air Management District (California) such as Azusa, the previous Federal limit of 0.08 ppm O_3 was exceeded about 1350 hours per year in 1974. A "management air quality standard" (Ref. 6-4) might specify that the number of hours per year should be reduced in stages from 1350 hours per year to, for example, 100 hours per year by 1987. This approach would permit statistically robust feedback from the observed frequency of specified concentrations to the predicted concentrations. Air quality managers could then alter their strategies by accelerating or decelerating the planned rates of reductions in emissions of specific pollutants.

This "management standards" approach based on technical, economic and social feasibility would set up practical milestones en route to cleaner air. A reduction in RHC and NO_x emissions by a factor of 2, in the South Coast Air Basin, over the period 1980-87, could reduce the number of hours per year that the Federal level of 0.12 ppm is exceeded by a factor of 10. A reduction of this magnitude is feasible in the South Coast Air Quality Management District (Ref. 6-5). The concept of a gradual but significant reduction in number of hours per year that the Federal level is exceeded may be extended, of course, to all AQMD's that presently violate the existing Federal standard.

6.4.2 Uncertainties in Transportation Emission Sources

The status of air quality management has been examined considering the uncertainties and expected errors in:

(1) Mobile sources emissions based on surveillance and factory testing.

(2) Emissions caused by measurement variability.

(3) The model estimates for stationary sources using available averages and yearly measured data to model peaks and hourly rates.

(4) Travel demand forecasting using transportation network models.

Based on the identified uncertainties in both the factory and surveillance test results, it is important to estimate useful confidence ranges for motor vehicles as a function of model year or mileage for each emission pollutant. The data taken in the CARB surveillance tests indicate a large scatter. It should be noted that the CARB tests had the benefit of using knowledge based on earlier tests and simulating conditions derived from the EPA Emissions Factor Test Program. One major concern is the lack of correlation in the data. This lack of correlation may be due to the single test per car that is run for each of the mileage intervals. The factory testing bears this out because of the large variability in data from repeated tests on the same car under the same conditions. The factory test data showed a +33% variability at the 95% confidence limit for CO test results due to the combined effects of measurement error and vehicle variability.

The large demonstrated uncertainty (% variability) in factory test data is supported by the large deviations shown from surveillance test data. In one case, the results from Ref. 6-6 for a sample of 53 1975 model year California cars showed a mean of 2.3 g/mi and a standard deviation of 1.7 g/mi of HC exhaust emission. However, the same set of tests for 1979 cars yielded much smaller value for both the mean (0.5 g/mi) and the standard deviation (p.4 g/mi). Two additional important points obtained from the examination of this data are:

(1) the use of Gaussian statistics to analyze the data is misleading as shown in these cases where the standard deviation is approximately equal to or greater than the mean, and (2) the percent of cars passing the tests was significantly improved by the newer models. This improvement in reducing emissions as demonstrated by the new cars may have a significant impact on the effectiveness of I/M over the next 5-10 years.

Two recent regulations issued by the EPA will assist motorists in states that have adopted I/M as part of their air quality plans (Ref. 6-8). The regulations established an EPA approved emissions testing method of cars to determine if light vehicles after 1981 exceed tailpipe standards. This short test, which is used to screen in-use "warm" cars for further evaluation or return to manufacturer for repairs, will place the burden on the manufacturer to have better control and to minimize early emission control system problems.

6.5 SUMMARY

In this section, the findings of the report are summarized. Some of these findings have led to conclusions, and some conclusions appear to warrant recommendations. The conclusions and recommendations are also found in this section.

6.5.1 Findings

A wide variety of findings have been determined in this report. In this summary we shall concentrate only on the major findings. We have found that the uncertainties in emission inventory are uncertain for both stationary sources and for mobile sources. We have found that the instrumentation available to measure both oxides of nitrogen and non-methane hydrocarbons has considerable errors in measurement. We have found that the air quality models examined have errors which range from +100% to -70% for linear rollback to ±50% for photochemical dispersion modeling. It should also be noted that these error estimates assume that the errors in the emission inventories are negligible compared to the errors in the model per se. This assumption does not appear to be warranted.

We have found that introducing increased complexity in a model does not necessarily decrease the total error. In fact, as complexity increases beyond that warranted by the state of measurements, the overall error produced by the model increases. We have found that it is very difficult to get unbiased estimates of either precision of air quality models or their accuracy. We have found that there may be a need to reassess non-methane hydrocarbon reduction strategies since some hydrocarbons appear to be much more reactive than others. The inspection and maintenance programs for automobiles have been found to produce significant reductions in emissions.

6.5.2 Conclusions

We have concluded that it would be quite beneficial to air quality managers if they knew the uncertainties in the emission inventories. This knowledge could be used as a basis for improving the emission inventories both in terms of estimating the most likely value and also in formulating strategies to reduce the uncertainties. This estimate of uncertainties could focus the efforts of the individual control agencies toward reducing the uncertainties from year to year. We have concluded that the measurement uncertainties in both NO_x and non-methane hydrocarbon are large enough to cause significant uncertainties in the use of air quality models.

We have concluded that it would be quite useful to have a study which would determine the total error for a given model as a function of complexity so that a given air quality management district could choose a model of the complexity which would minimize its errors in terms of air quality management. We have concluded that there is a need to specify both the precision and the accuracy for air quality models. At present, we cannot find these specified in an unbiased manner. We have concluded that measurements of reasonable further progress are hampered by the lack of estimates of the uncertainties associated with the emission inventories. We have concluded that automobile inspection maintenance can produce significant reductions in air emissions for the three regions under consideration.

The present Federal ambient air quality standard for oxidant expressed in terms of a maximum hourly average concentration not to be exceeded more than one day per year is based on effects on human health. We have concluded that the uncertainties associated with the tools presently available for air quality management render these tools inadequate to accurately predict such a rare event. We recommend that serious consideration be given by the EPA to replacing the present procedure by a "management standards" approach that is statistically robust. This approach would set up milestones on the way to clean air in terms of hours per year that the Federal standard is exceeded in those AQMD's that presently violate the Federal standard by a large margin. For example, a reduction of 50% in the number of hours per year that the Federal level is exceeded might be mandated for the period 1979-1983. An additional reduction of 75% in the remaining number of hours per year that the Federal level is exceeded might be mandated for the period 1983-1987. This approach would permit essential feedback to air quality managers responsible for planning the required reductions in emissions of specific pollutants.

REFERENCES

6-1 National Commission on Air Quality, Los Angeles Regional Study - SIP Process Review, Executive Summary by Systems Applications, Inc., ACUREX Corporation, and Energy and Environmental Analysis, Inc., Feb. 1980, Draft Report 80-46/EE, pg. 4-3.

6-2 "Discussion Paper--An Assessment of the Uncertainty in the Analysis of Recommended AQMP Measures," Southern California Association of Governments, Paul Wuebben to T. English, Dec. 1979.

6-3 "An Analysis of the Impact of the Vehicular Inspection/ Maintenance Program on Ambient Carbon Monoxide," Division of Environmental Health Services, Arizona Department of Health Services, Nov. 1979, pg. 7.

6-4 Eschenroeder, A., "Mobile Source Emissions Reductions in the South Coast Air Basin, #2 Biases in Emissions Estimates and Their Effects on Air Quality Projections," Southern California Edison R&D Series 79-RD-138, Oct. 1979.

6-5 Trijonis, J., Peng, T., McRae, G., and Lees, L., "Emissions and Air Quality Trends in the South Coast Air Basin," Environmental Quality Laboratory, California Institute of Technology, Pasadena, California Jan. 1979, pg. 33.

6-6 Air Quality Management Plan, SCAQMD and SCAG, Jan. 1979, pg. VIII-33.

6-7 Moore, L., and Unger, D.C., Jr., "A Study of Exhaust Emissions from 1975 to 1979 Model Year Passenger Cars in Los Angeles," EPA-460/3-79-004, Automotive Environmental Systems Inc., 7300 Bolsa Ave., Westminster, CA 92683, July 1979.

6-8 Environmental News, EPA Issues New Regulations on Auto Inspections, USEPA, Press Office (A-107) Washington, D.C. 20460, May 22, 1980.

APPENDIX A. NEW VEHICLE STANDARDS SUMMARIES

Table A-1. Passenger Cars

The following California standards, up through 1979, and federal standards, up through 1974, apply only to gasoline-powered passenger cars(1). Federal standards for 1975 and later and California standards for 1980 and later apply to both gasoline and diesel powered passenger cars.

Year	Standard	Test Procedure	Hydro Carbons	Carbon Monoxide	Oxides of Nitrogen
Prior to controls	none	7-mode	850 ppm	3.4%	1000 ppm
		7-mode	11 gm/mi	80 gm/mi	4 gm/mi
		CVS-75	8.8 gm/mi	87.0 gm/mi	3.6 gm/mi
1966-67	Calif.	7-mode	275 ppm	1.5%	no std.
1968-69	Calif. & federal	7-mode			
		50-100 CID	410 ppm	2.3%	no std.
		101-140 CID	350 ppm	2.0%	no std.
		over-140 CID	275 ppm	1.5%	no std.
1970	Calif. & federal	7-mode	2.2 gm/mi	23 gm/mi	no std.
1971	Calif.	7-mode	2.2 gm/mi	23 gm/mi	4.0 gm/mi
	federal	7-mode	2.2 gm/mi	23 gm/mi	no std.
1972	Calif.	7-mode or	1.5 gm/mi	23 gm/mi	3.0 gm/mi
		CVS-72	3.2 gm/mi	39 gm/mi	3.2 gm/mi(2)
	federal	CVS-72	3.4 gm/mi	39 gm/mi	no std.
1973	Calif.	CVS-72	3.2 gm/mi	39 gm/mi	3.0 gm/mi
	federal	CVS-72	3.4 gm/mi	39 gm/mi	3.0 gm/mi
1974	Calif.	CVS-72	3.2 gm/mi	39 gm/mi	2.0 gm/mi
	federal	CVS-72	3.4 gm/mi	39 gm/mi	3.0 gm/mi
1975	Calif.	CVS-75	0.9 gm/mi(3)	9.0 gm/mi	2.0 gm/mi
	federal	CVS-75	1.5 gm/mi	15 gm/mi	3.1 gm/mi
1976	Calif.	CVS-75	0.9 gm/mi(3)	9.0 gm/mi	2.0 gm/mi
	federal	CVS-75	1.5 gm/mi	15 gm/mi	3.1 gm/mi
1977	Calif.	CVS-75	0.41 gm/mi	9.0 gm/mi	1.5 gm/mi
	federal	CVS-75	1.5 gm/mi	15 gm/mi	2.0 gm/mi

Table A-1. Passenger Cars (Continuation 1)

Year	Standard	Test Procedure	Hydro Carbons	Carbon Monoxide	Oxides of Nitrogen
1978	Calif.	CVS-75	0.41 gm/mi	9.0 gm/mi	1.5 gm/mi
	federal	CVS-75	1.5 gm/mi	15 gm/mi	2.0 gm/mi
1979	Calif.	CVS-75	0.41 gm/mi	9.0 gm/mi	1.5 gm/mi
	federal	CVS-75	1.5 gm/mi	15 gm/mi	2.0 gm/mi

Year	Standard	Test Procedure	Non-Methane Hydro-Carbons (4)	Carbon Monoxide	Oxides of Nitrogen
1980	Calif.	CVS-75	0.39 gm/mi	9.0 gm/mi	1.0 gm/mi
	100,000 Option 1		0.39 (.41)	9.0 gm/mi	1.5 gm/mi
	100,000 Option 2		0.46 gm/mi	10.6 gm/mi	1.5 gm/mi
	federal		(0.41 gm/mi)	7.0 gm/mi	2.0 gm/mi
1981	(5)Calif.				
	(A)	CVS-75	(0.41 gm/mi)	3.4 gm/mi	1.0 gm/mi
	(B)		0.39 (.41)	7.0 gm/mi	0.7 gm/mi
	100,000 Option 1		0.41 gm/mi	3.4 gm/mi	1.5 gm/mi
	100,000 Option 2		0.46 gm/mi	4.0 gm/mi	1.5 gm/mi
	federal		(0.41 gm/mi)	3.4 gm/mi	1.0 gm/mi
1982	Calif.				
	(A)	CVS-75	0.39 (.41)	7.0 gm/mi	0.4 gm/mi
	(B)		0.39 (.41)	7.0 gm/mi	0.7 gm/mi
	100,000 Option 1		0.39 (.41)	7.0 gm/mi	1.0 gm/mi
	100,000 Option 2		0.46 gm/mi	8.3 gm/mi	1.0 gm/mi
	federal		(0.41 gm/mi)	3.4 gm/mi	1.0 gm/mi
1983	Calif.	CVS-75	0.39 (.41)	7.0 gm/mi	0.4 gm/mi
	100,000 (All)		0.39 (.41)	7.0 gm/mi	1.0 gm/mi
	100,000 (All)		0.46 gm/mi	8.3 gm/mi	1.0 gm/mi
	federal		(0.41 gm/mi)	3.4 gm/mi	1.0 gm/mi

ARB Fact Sheet 6 (Revised 9/20/79)

Table A-1. Passenger Cars (Continuation 2)

(1) Passenger car as defined in Title 13 of the California Administrative Code means any motor vehicle designed primarily for transportation of persons having a capacity of twelve persons or less.

(2) Hot 7-mode test.

(3) Hydrocarbon emissions from 1975-76 limited production vehicles may not exceed 1.5 gm/mi.

(4) Hydrocarbon standards in parentheses indicate total hydrocarbons. When applicable, manufacturers may elect to certify vehicles to either the non-methane or total hydrocarbon standards.

(5) For the 1981 model year manufacturers may choose options A or B separately for their gasoline and diesel product lines. The option chosen in 1981 must be retained for the 1982 model-year.

50,000 mile certification applies to all standards not listed at 100,000 miles.

100,000 mile options: Option 1 standards refer to the projected 50,000 mile emissions for hydrocarbons and carbon monoxide while Option 2 standards refer to the projected 100,000 mile emissions for these pollutants. NOx emission standards for both options refer to the 100,000 mile projected emissions.

For vehicles with evaporative emissions values below 1.0 gm/test, and adjustment to the exhaust hydrocarbon emissions standard may be granted by the Executive Office (100,000 mile option only).

gm/mi - grams per mile.
CVS-72 - constant volume sample cold start test.
CVS-75 - constant volume sample test which includes cold and hot starts.
7-mode - 137 second driving cycle test.
ppm - parts per million.

Other Requirements

Effective 1978, evaporative emission standards are 6.0 grams per SHED test for the 1978-79 model years and 2.0 grams per SHED test for 1980 and later model years.

Table A-1. Passenger Cars (Continuation 3)

Beginning with the 1980 model-year, the maximum projected emissions of oxides of nitrogen measured on the federal Highway Fuel Economy Test shall be no greater than 1.33 times the applicable passenger car oxides of nitrogen standards shown above.
SHED Test (Sealed Housing Evaporative Determination) - A method for measuring evaporative emissions from motor vehicles.

Table A-2. Light-Duty Trucks

The following California standards, up through 1977, and federal standards, up through 1975, apply only to gasoline-powered light-duty trucks(1). Federal standards for 1975 and later and California standards for 1978 and later apply to both gasoline and diesel powered light-duty trucks.

Year	Standard	Test Procedure	Hydro-Carbons	Carbon Monoxide	Oxides of Nitrogen
1966-67	Calif.	7-mode	275 ppm	1.5%	no std.
1968-69	Calif. & federal	7-mode			
		50-100 CID	410 ppm	2.3%	no std.
		101-140 CID	350 ppm	2.0%	no std.
		over-140 CID	275 ppm	1.5%	no std.
1970	Calif. & federal	7-mode	2.2 gm/mi	23 gm/mi	no std.
1971	Calif.	7-mode	2.2 gm/mi	23 gm/mi	4.0 gm/mi
	federal	7-mode	2.2 gm/mi	23 gm/mi	no std.
1972	Calif.	7-mode or	1.5 gm/mi	23 gm/mi	3.0 gm/mi
		CVS-72	3.2 gm/mi	39 gm/mi	3.2 gm/mi(2)
	federal	CVS-72	3.4 gm/mi	39 gm/mi	no std.
1973	Calif.	CVS-72	3.2 gm/mi	39 gm/mi	3.0 gm/mi
	federal	CVS-72	3.4 gm/mi	39 gm/mi	3.0 gm/mi
1974	Calif.	CVS-72	3.2 gm/mi	39 gm/mi	2.0 gm/mi
	federal	CVS-72	3.4 gm/mi	39 gm/mi	3.0 gm/mi
1975	Calif.	CVS-75	2.0 gm/mi	20 gm/mi	2.0 gm/mi
	federal	CVS-75	2.0 gm/mi	20 gm/mi	3.1 gm/mi
1976	Calif.	CVS-75	0.9 gm/mi	17 gm/mi	2.0 gm/mi
	federal	CVS-75	2.0 gm/mi	20 gm/mi	3.1 gm/mi
1977	Calif.	CVS-75	0.9 gm/mi	17 gm/mi	2.0 gm/mi
	federal	CVS-75	2.0 gm/mi	20 gm/mi	3.1 gm/mi
1978	Calif.	CVS-75	0.9 gm/mi	17 gm/mi	2.0 gm/mi
	federal	CVS-75	2.0 gm/mi	20 gm/mi	3.1 gm/mi
1979	Calif.(3)	CVS-75	0.41 gm/mi	9.0 gm/mi	1.5 gm/mi
	Calif.(4)	CVS-75	0.50 gm/mi	9.0 gm/mi	2.0 gm/mi
1979	federal(5)	CVS-75	1.7 gm/mi	18 gm/mi	2.3 gm/mi

Table A-2. Light-Duty Trucks (Continuation 1)

Year	Standard	Test Procedure	Non-Methane Hydrocarbons(6)	Monoxide	Oxides of Nitrogen
1980	Calif.(3)	CVS-75	0.39 gm/mi (0.41 gm/mi)	9.0 gm/mi	1.5 gm/mi
	Calif.(4)	CVS-75	0.50 gm/mi (0.50 gm/mi)	9.0 gm/mi	2.0 gm/mi
	federal		(1.7 gm/mi)	18.0 gm/mi	2.3 gm/mi
1981	Calif.(3)	CVS-75	0.39 (.41)	9.0 gm/mi	1.0 gm/mi
	100,000 Option 1		0.39 gm/mi	9.0 gm/mi	1.5 gm/mi
	100,000 Option 2		0.46 gm/mi	10.6 gm/mi	1.5 gm/mi
	Calif.(4)		0.50 (.50)	9.0 gm/mi	1.5 gm/mi
	100,000		0.50 (.50)	9.0 gm/mi	2.0 gm/mi
	federal		(1.7 gm/mi)	18.0 gm/mi	2.3 gm/mi
1982	Calif.(3)	CVS-75	0.39 (.41)	9.0 gm/mi	1.0 gm/mi
	100,000 Option 1		0.39 (.41)	9.0 gm/mi	1.5 gm/mi
	100,000 Option 2		0.46 gm/mi	10.6 gm/mi	1.5 gm/mi
	Calif.(4)		0.50 (.50)	9.0 gm/mi	1.5 gm/mi
	100,000		0.50 (.50)	9.0 gm/mi	2.0 gm/mi
	federal		(1.7 gm/mi)	18.0 gm/mi	2.3 gm/mi
1983 & Sub.	Calif.(3)	CVS-75	0.39 (.41)	7.0 gm/mi	0.4 gm/mi
	100,000 Option 1		0.39 (.41)	7.0 gm/mi	1.0 gm/mi
	100,000 Option 2		0.46	10.6 gm/mi	1.0 gm/mi
	Calif.(4)		0.50 (.50)	9.0 gm/mi	1.0 gm/mi
	100,000		0.50 (.50)	9.0 gm/mi	1.5 gm/mi
1983 1984	federal		(0.99 gm/mi)	9.4 gm/mi	2.3 gm/mi
1985 & later	federal		(0.99 gm/mi)	9.4 gm/mi	1.4 gm/mi

(1) Light-duty trucks as defined by Title 13 of the California Administrative Code means any motor vehicle rated at 6000 lbs.GVW or less which is designed primarily for purposes of transportation of property or is a derivative of such vehicle, or is available with special features enabling off-street or off-highway operation and use.

(2) Hot 7-mode test.

(3) 0-3999 pounds equivalent inertia weight (curb weight plus 300 pounds).

(4) 4000-6000 pounds equivalent inertia weight.

(5) Effective 1979, federal LDT classification will be extended to 8500 pounds GVW.

Table A-2. Light-Duty Trucks (Continuation 2)

(6) Hydrocarbon standards in parentheses indicate total hydrocarbon. When applicable, manufacturers may elect to certify vehicles to either the non-methane or total hydrocarbon standards.

50,000 miles certification applies to all standards not listed at 100,000 miles.

100,000 mile options: Option 1 standards refer to the projected 50,000 mile emissions for hydrocarbons and carbon monoxide while Option 2 standards refer to the projected 100,000 mile emissions for these pollutants. NOx emission standards for both options refer to the 100,000 mile projected emissions.

For vehicles with evaporative emissions values below 1.0 gm/test, an adjustment to the exhaust hydrocarbon emissions standard may be granted by the Executive Officer (100,000 mile option only).

gm/mi - grams per mile.
CVS-72 - constant volume sample cold start test.
CVS-75 - constant volume sample which includes cold and hot starts.
7-mode - 137 second driving cycle test.
ppm - parts per million

Other Requirements:

Effective 1978, evaporative emission standards are 6.0 grams per SHED test for 1978-79 model-year and 2.0 grams per SHED test for 1980 and later model-year.

Beginning with the 1981 model-year, the maximum projected emissions of oxides of nitrogen measured on the federal Highway Fuel Economy Test shall be no greater than 2.0 times the applicable light-duty truck oxides of nitrogen standards shown above.

SHED Test (Sealed Housing Evaporative Determination) - A method for measuring evaporative emissions from motor vehicles.

Table A-3. Medium-Duty Vehicles

The following California standards, up through 1972, and federal standards, up through 1973, apply only to gasoline-powered medium-duty vehicles(1). California standards for 1973 and later year and federal standards for 1974 and later apply to both gasoline and diesel powered medium-duty vehicles.

Year	Standard	Test Procedure	Non-Methane Hydrocarbons(5)	Carbon Monoxide	Oxides of Nitrogen
1969-77	Calif.	See HEAVY-DUTY STANDARDS for 1969-1977			
1970-78	federal	See HEAVY-DUTY STANDARDS for 1970-1978			
1978(2)	Calif.	CVS-75	(0.9 gm/mi)	17.0 gm/mi	2.3 gm/mi
1979	Calif.	CVS-75	(0.9 gm/mi)	17.0 gm/mi	2.3 gm/mi
	federal	See LIGHT-DUTY TRUCK STANDARDS for 1979 and later			
1980	Calif.	CVS-75	(0.9 gm/mi)	17.0 gm/mi	2.3 gm/mi
	federal		(1.7 gm/mi)	18.0 gm/mi	2.3 gm/mi
1981	Calif.(3)	CVS-75	0.39 (.41)	9.0 gm/mi	1.0 gm/mi
	100,000 Option 1		0.39 gm/mi	9.0 gm/mi	1.5 gm/mi
	100,000 Option 2		0.46 gm/mi	10.6 gm/mi	1.5 gm/mi
	Calif.(4)		0.50 (.50)	9.0 gm/mi	1.5 gm/mi
	100,000		0.50 (.50)	9.0 gm/mi	2.0 gm/mi
	Calif.(5)		0.60 (.60)	9.0 gm/mi	2.0 gm/mi
	100,000		0.60 (.60)	9.0 gm/mi	2.3 gm/mi
	federal		(1.7 gm/mi)	18.0 gm/mi	2.3 gm/mi
1982	Calif.(3)	CVS-75	0.39 (.41)	9.0 gm/mi	1.0 gm/mi
	100,000 Option 1		0.39 (.41)	9.0 gm/mi	1.5 gm/mi
	100,000 Option 2		0.46 gm/mi	10.6 gm/mi	1.5 gm/mi
	Calif.(4)		0.50 (.50)	9.0 gm/mi	1.5 gm/mi
	100,000		0.50 (.50)	9.0 gm/mi	2.0 gm/mi
	Calif.(5)		0.60 (.60)	9.0 gm/mi	2.0 gm/mi
	100,000		0.60 (.60)	9.0 gm/mi	2.3 gm/mi
	federal		(1.7 gm/mi)	18.0 gm/mi	2.3 gm/mi
1983 & Sub.	Calif.(3)	CVS-75	0.39 (.41)	7.0 gm/mi	0.4 gm/mi
	100,000 Option 1		0.39 (.41)	7.0 gm/mi	1.0 gm/mi
	100,000 Option 2		0.46 gm/mi	10.6 gm/mi	1.0 gm/mi
	Calif.(4)		0.50 (.50)	9.0 gm/mi	1.0 gm/mi
	100,000		0.50 (.50)	9.0 gm/mi	1.5 gm/mi
	Calif.(5)		0.60 (.60)	9.0 gm/mi	1.5 gm/mi
	100,000		0.60 (.60)	9.0 gm/mi	2.0 gm/mi
1983-84	federal		(0.99 gm/mi)	9.4 gm/mi	2.3 gm/mi
1985 & later	federal		(0.99 gm/mi)	9.4 gm/mi	1.4 gm/mi

Table A-3. Medium-Duty Vehicles (Continuation 1)

(1) Medium-duty vehicles as defined in Title 13 of the California Administrative Code means any heavy-duty vehicles having a manufacturer's gross vehicle weight rating of 8500 ounds or less (Manufacturers may elect to certify medium-duty vehicles up to 10,000 pounds GVW).

(2) 0-3999 pounds equivalent inertia weight (curb weight plus 300 pounds).

(3) 4000-6000 pounds equivalent inertia weight.

(4) 6001-8500 (or 10,000) pounds equivalent inertia weight.

(5) Hydrocarbon standards in parentheses indicate total hydrocarbon. When applicable, manufacturers may elect to certify vehicles to either the non-methane or total hydrocarbon standards.

50,000 miles certification applies to all standards not listed at 100,000 miles.

100,000 mile options: Option 1 standards refer to the projected 50,000 mile emissions for hydrocarbons and carbon monoxide while Option 2 standards refer to the projected 100,000 mile emissions for these pollutants. NOx emission standards for both options refer to the 100,000 mile projected emissions.

For vehicles with evaporative emissions values below 1.0 gm/test, an adjustment to the exhaust hydrocarbon emissions standard may be granted by the Executive Officer (100,000 mile option only).

gm/mi - grams per mile
CVS-75 - constant volume sample test which includes cold and hot starts.

Other Requirements:

Effective 1978, evaporative emission standards are 6.0 grams per SHED test for the 1978-79 model-years and 2.0 grams per SHED test for 1980 and later model-years.

Beginning with the 1981 model-year, the maximum projected emissions of oxides of nitrogen measured on the federal Highway Fuel Economy Test shall be no greater than 2.0 times the applicable medium-duty vehicle oxides of nitrogen standards shown above.

SHED Test (Sealed Housing Evaporative Determination) - A method for measuring evaporative emissions from motor vehicles.

Table A-4. Heavy-Duty Engines and Vehicles
(Diesel (1) and Gasoline)

The following is a summary of heavy-duty engine and vehicle(2) standards adopted by both the California Air Resources Board and federal Environmental Protection Agency.

Year	Standard	Hydro-Carbons	Carbon Monoxide	Oxides of Nitrogen	Hydrocarbons & Oxides of Nitrogen
1969-71(3)	Calif.	275 ppm	1.5%	no std.	no std.
1970-72(4)	federal	275 ppm	1.5%	no std.	no std.
1972	Calif.	180 ppm	1.0%	no std.	no std.
1973-74	Calif.	no std.	40 gm/BHP-hr	no std.	16 gm/BHP-hr
1973-78	federal	no std.	40 gm/BHP-hr	no std.	16 gm/BHP-hr
1975-76	Calif.	no std.	30 gm/BHP-hr	no std.	10 gm/BHP-hr
1977-78	Calif. or	no std.	25 gm/BHP-hr	no std.	5 gm/BHP-hr
	Calif.	1.0 gm/BHP-hr	25 gm/BHP-hr	7.5 gm/BHP-hr	no std.
1979(5)	Calif. or	1.5 gm/BHP-hr	25 gm/BHP-hr	7.5 gm/BHP-hr	no std.
	Calif. or	no std.	25 gm/BHP-hr	no std.	5 gm/BHP-hr
	Calif.	1.0 gm/BHP-hr	25 gm/BHP-hr	7.5 gm/BHP-hr	no std.
1979-1982	federal or	1.5 gm/BHP-hr	25 gm/BHP-hr	no std.	10 gm/BHP-hr
gasoline	federal	no std.	25 gm/BHP-hr	no std.	5 gm/BHP-hr
diesel		1.5 gm/BHP-hr	25 gm/BHP-hr	10	-
		-	25 gm/BHP-hr	-	5 gm/BHP-hr
1980-82	Calif. or	1.0 gm/BHP-hr	25 gm/BHP-hr	no std.	6.0 gm/BHP-hr
	Calif.	no std.	25 gm/BHP-hr	no std.	5 gm/BHP-hr
1983 &	Calif.	0.5 gm/BHP-hr	25 gm/BHP-hr	no std.	4.5 gm/BHP-hr
later*	federal	1.3 gm/BHP-hr	15.5 gm/BHP-hr	10.7	-

(1) The above standards apply to diesel engines and vehicles sold in California on or after January 1, 1973 and nationwide on or after January 1, 1974.

(2) These standards apply to motor vehicles having a manufacturer's GVW rating of over 6000 pounds, excluding passenger cars and 1978 and later medium-duty vehicles.

(3) Applies to vehicles manufactured on or after January 1, 1969.

(4) Applies to vehicles manufactured on or after January 1, 1970.

(5) For 1979 only, manufacturer using heated flame ionization detection (HFID) method of measuring hydrocarbons must meet the 1.5 gm/BHP-hr standard; whereas manufacturers using non-dispersive infrared (NDIR) method of measuring hydrocarbons must meet the 1.0 gm/BHP-hr standard. Both standards are equivalent in stringency. Manufacturers may use either HFID or NDIR in meeting the combined hydrocarbon and oxides of nitrogen standard of 5 gm/BHP-hr. After 1979, manufacturers are required to use HFID.

* There are no federal standards for 1983 - these figures apply to 1984.

Table A-4. Heavy-Duty Engines and Vehicles
(Diesel (1) and Gasoline)
(Continuation 1)

gm/BHP-hr - grams per brake horsepower - hour
ppm - parts per million.

Additional Requirements

Effective 1978, evaporative emission standards are 6.0 grams per SHED test for the 1978-79 model-years and 2.0 grams per SHED test for 1980 and later model years.

SHED Test (Sealed Housing Evaporative Determination) A method for measuring evaporative emissions from motor vehicles.

Table A-5. Motorcycles

The following is a summary of motorcycle(1) standards adopted by both the California Air Resources Board and the federal Environmental Protection Agency.

Year	Standard	Displacement(2)	Hydrocarbons	Carbon Monoxide
1978-79	Calif. & federal	50-169	5.0 gm/km	17 gm/km
		170-749	5.0 + 0.0155(D-170) gm/km (3)	17 gm/km
		750 & larger	14 gm/km	17 gm/km
1980-81	Calif.	All (50 & larger)	5.0 gm/km	12 gm/km
1980 & later	federal	All (50 & larger)	5.0 gm/km	12 gm/km
1982 & later	Calif.	All (50 & larger)	1.0 gm/km	12 gm/km

(1) Any motor vehicle other than a tractor having a seat or saddle for the use of the rider and designed to travel on not more than three wheels in contact with the ground and weighing less than 1500 pounds, except that four wheels may be in contact with the ground when two of the wheels function as a sidecar.

(2) Displacement shown in cubic centimeters.

(3) Motorcycle Hydrocarbon Formula

D = engine displacement in cubic centimeters
e.g., 300 cc engine; standard = (300-170) X .0155 + 5.0
= 7.0 gm/km

gm/km - grams per kilometer

Additional Requirement

Effective 1983, evaporative emission standards are 6.0 grams per SHED test for 1983-84 model-years and 2.0 grams per SHED test for 1985 and later model-year.

SHED Test (Sealed Housing Evaporative Determination) - A method for measuring evaporative emissions from motorcycles.

APPENDIX B. TESTING AND MODELING

B.1 TEST METHODS AND PROCEDURES

Test methods and procedures pertaining to both regulated emissions and fuel economy of LDV have been assessed by the National Academy of Sciences (NAS) through its Committee on Motor Vehicle Emissions (CMVE) (Ref. B-1). Their findings were that the exhaust reliability and reproducibility of emissions and fuel economy measurements for LDV were strongly influenced by the specified test method and procedures. In particular, the CVS-CH test methods and the 1975 Federal Test Procedure (FTP) were evaluated. The CVS-CH test cycle was evaluated to determine how closely it approximates actual conditions in an urban area and to estimate variations in emission results during the tests.

The intent of this examination of test methods and procedures is to help DOT develop a detailed data base for the quantitative assessment of the factors that influence control of emissions and fuel economy. The information will also be helpful in the development of standardized test methods and procedures that adequately demonstrate emission control and fuel economy.

Of particular interest was the conclusion that based on a comparison with five city composite cycle tests performed under the CAPE-10 project, the CVS-CH test method using the Urban Dynamometer Driving Cycle (UDDS) is a sufficiently accurate representation of urban driving patterns to be used for purposes of determining LDV exhaust emissions (Ref. B-1, p. 2).

A brief discussion of the 1975 FTP, the 2-speed idle test and the Federal three mode tests are presented below because much of the data is obtained through this type of test.

B.1.1 1975 Federal Test Procedure

The 1975 FTP is described in 40 FR 126. The 1975 FTP requires a 12-hour soak at normal room temperature before starting the vehicle. The first five cycles, from time zero to 505 sec, are referred to as the cold-start phase. Cycles 6 through 18 (505 to 1372 sec) are called the stabilized phase. At the end of cycle 18, the engine is switched off for 10 min. The first five cycles are then repeated (i.e., cycles 19 through 23); this third phase is called the hot-start. To obtain the official FTP emissions and fuel economy values, the data from the cold-start and hot-start phases are weighted by factors of 0.43 and 0.57, respectively, to reflect estimated proportions of in-use cold and hot starts. A detailed outline of the 1975 FTP is described in Ref. B-2.

B.1.2 Two-Speed Idle Test

The two-speed idle test consists of volumetric sampling of undiluted exhaust emissions during two steady-state operating conditions with the hood open and the cooling fan on. The first operating mode is at 2500 engine rpm with the transmission in neutral. The second mode is at normal idle with the transmission in neutral also.

Two-speed idle tests are preceded by a 6-min soak period (3 min with engine off and 3 with the engine idling). At the end of the soak period the vehicle is operated for a maximum of 3 min at 2500 rpm and then at idle speed. At each speed, equilibrium of engine speed and the CO, HC and NO analyzer output meters are maintained for 30 sec before the readings are recorded. CO, HC and NO are measured and reported in percent CO, ppm/hexane and ppm NO respectively. The rpm should be written on the trace for both operating conditions (Ref. B-3).

B.1.3 Federal Three-Mode Test

The Federal three-mode test procedure is preceded by a 6-min soak as described under the two-speed idle test. The test consists of three steady-state operating modes from which undiluted exhaust samples are taken. The dynamometer loads simulate the average power that occurs at the appropriate speed on the FTP, with all light duty vehicles being grouped into weight classes. At the end of the soak period the vehicle is operated for a maximum of 3 min at the mode-specified speed with the hood open and the cooling fan on. The inertia weight is set at 1750 lbs inertia. During this stabilization period the specified dynamometer load horsepower is set.

The test is performed in the order of high cruise, low cruise, and idle. The idle mode is sampled in both drive (automatic transmissions only) and in neutral. At each speed, equilibrium of vehicle speed and the CO, HC and NO_x analyzer meters are maintained for 30 sec before the readings were recorded. CO, HC and NO_x are measured and reported in percent CO, ppm/hexane and NO_x ppm respectively. The instruments used for measurement of undiluted HC, CO and NO_x emissions were the same as used for the two-speed idle test. Undiluted exhaust emissions are measured by the volumetric procedure for each mode (Ref. B-3).

B.2 FACTORY TESTING

Two major types of factory testing are performed on LDV to demonstrate either compliance with regulations or capability of the system to perform as designed. These tests are certification testing and developmental testing. Certification testing of LDV is performed as required by either Federal or State regulations for the current model year as defined in Federal Register FR Vol 42, No. 124. Developmental testing is performed generally by manufacturers to demonstrate that a particular emission control system will perform as designed and will meet or operate under the anticipated Federal Test Procedures. An

analysis of the results from developmental tests is more useful in this study because an examination of uncertainties and variabilities in test results for different test conditions and car mileage can be made. Certification testing is performed on new vehicles and practically all must pass depending on the regulations, or the car may go to a state with less constraining emission regulations.

B.2.1 Certification Testing

During this study no significant data base on the results of certification tests was obtained; however, results of certification testing are available and should be assessed. Examination of certification test results might lead to:

(1) Findings that could impact the procedures for certification of new vehicles.

(2) New vehicle tests that could be used to determine the nature of the analytic approximation to extremely low mileage cars, 100-1000 miles. This information may be useful in developing I/M requirements that could be the responsibility of dealers to repair components during initial use failure.

B.2.2 Developmental Testing

Engine control and driveability for vehicles are problems continuously being studied to minimize emissions and maximize fuel economy. Developmental testing underway as schematically shown in Fig. B-1 is producing results that demonstrate large variability among test vehicles. The emissions from these tests support the statistical inferences made in appraising vehicles in actual use and explain the large differences in surveillance test results. Engine control involves controlling the characteristics of the engine through feedback and driveability involves controllers' ability to deliver commanded values during all phases of the tests. As an example of engine control Dohner (Ref. B-4) demonstrates how the amounts of HC, CO and NO_x depend on the air-to-fuel ratio, spark advance, and exhaust gas recirculation devices under highly controlled and continuously recorded conditions. The results of Fig. B-2 show CO, HC and NO_x variations for each of these parameters using a 5.7 liter engine at 1600 rpm and 276 kPA BMEP (break mean effective pressure). These figures show a continuous response in the applied controls at the above speed-load conditions. The point is that significant variation in the emissions occur such as the highly nonlinear behavior shown for CO as the A/F ratio changes from just above 15 to 16. In order to show the influence of proper control using feedback, these tests were demonstrated to be repeatable (see Ref. B-4 p. 1315-1317, Fig. 4-1). The results of controlling the engine during the same speed-load conditions were then normalized after each iteration. The initial emission measurements are shown as unity on Fig. B-3. The decrease for HC emission was more than 50%. The

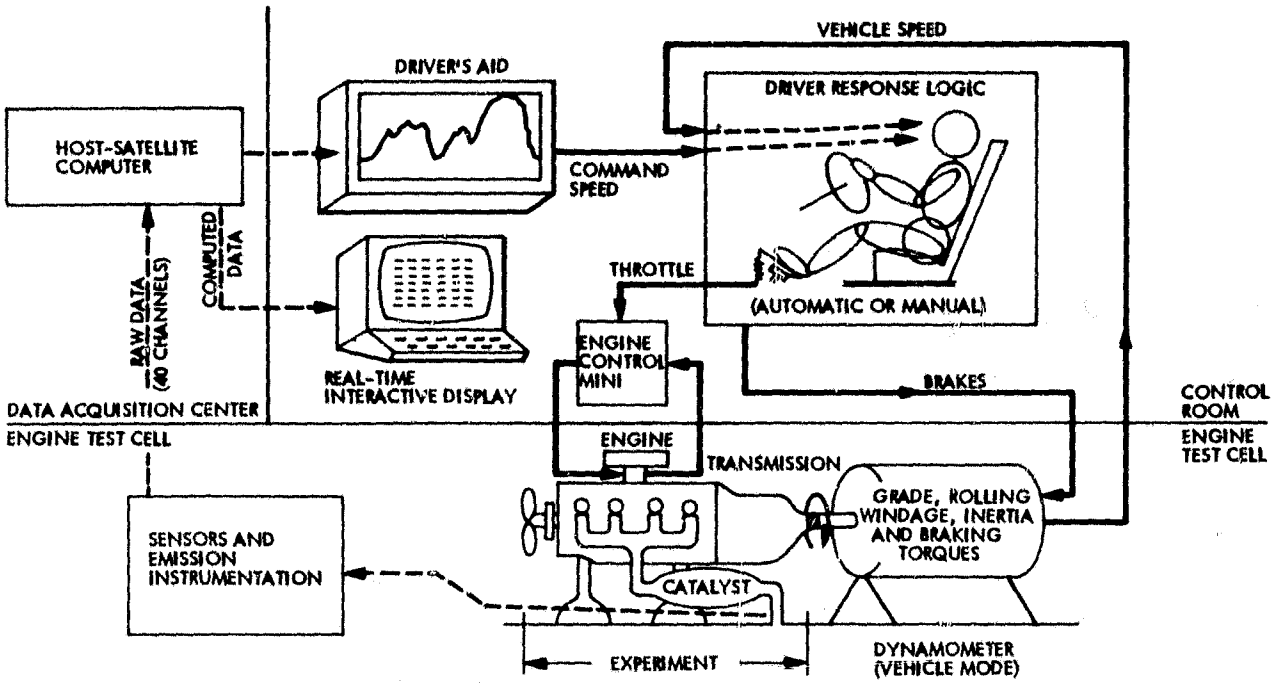


Fig. B-1. Experimental Test Cell Configuration (Ref. B-4)

NO_x was decreased by more than 35% and the CO, although decreased for each iteration, had remained below the constant value assigned at $\leq$9.0 g/mi (note that for this study HC and NO_x were constrained at $\leq$0.3 g/mi and $\leq$3.0 g/mi, respectively).

One interpretation of these results is that where control is imposed on the vehicle undergoing the FTP and the improvements are as shown in Fig. B-3, then the lack of both engine control and control in the laboratory in which to test, as in the surveillance tests, will lead to considerable variability in test results. The variability in emissions during actual operations by the driving public then could be even larger and should be accounted for in the results of the surveillance tests used to develop emission factors. The influence of surveillance variability is discussed in Section B.3. Variability during factory testing is discussed below.

B.2.3 Variability of Exhaust Emissions Measurements

Both systematic and random errors associated with vehicle variability, emission collection and measurement variability and environmental variables contribute to the test variability of a single vehicle. In addition, variations occur in emissions from a group of similar production vehicles (same year, same engine design and control system).

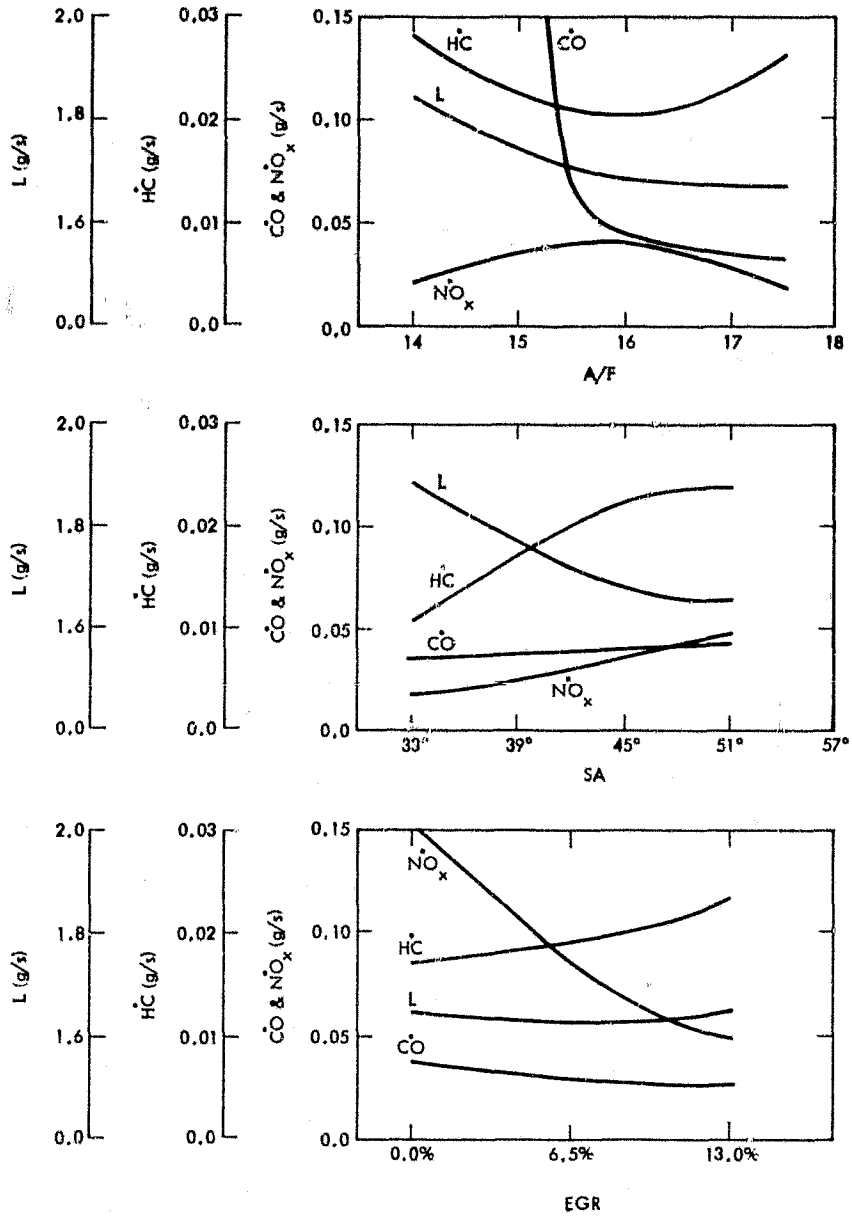


Fig. B-2. Fuel and Emission Rates at 1600 RPM and 276 kPa BMEP (Ref. B-4)

The sources of variability associated with the measured exhaust emissions are categorized in Fig. B-4. These factors affect the exhaust emissions variabilities within a given test cell, variability from cell-to-cell within a given laboratory, as well as laboratory-to-laboratory variability.

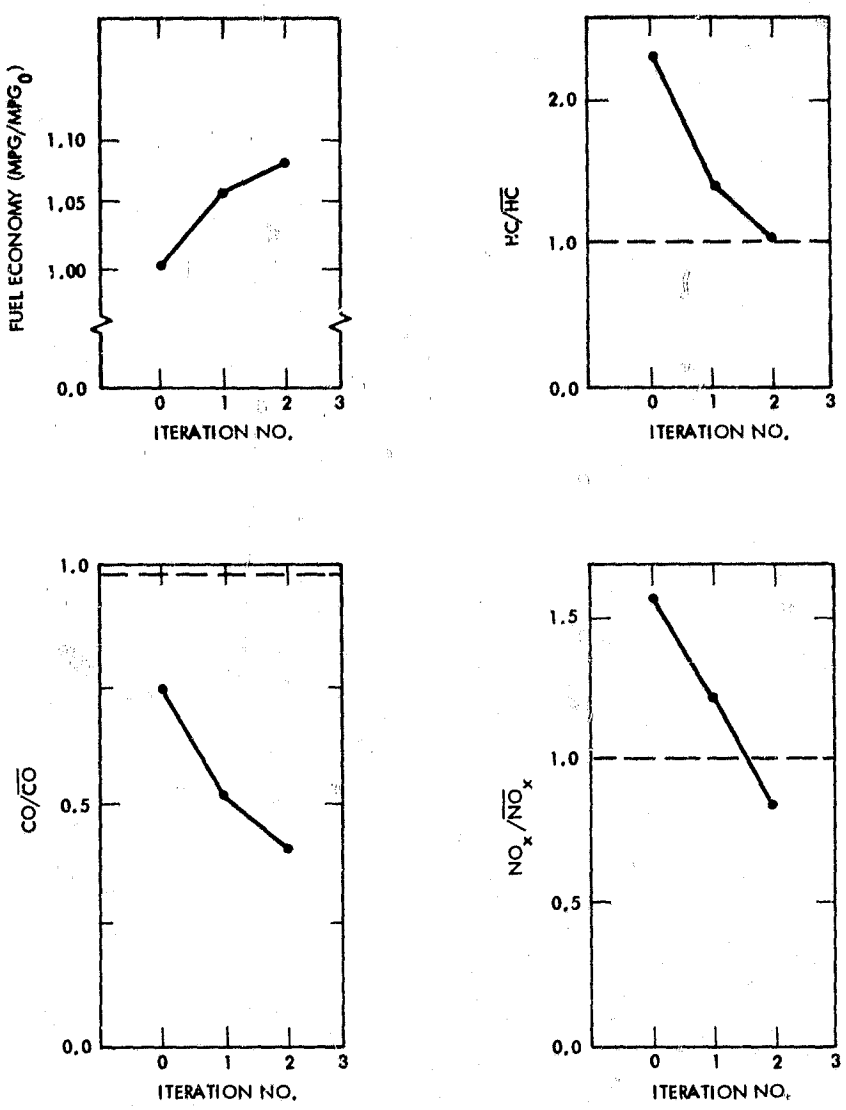


Fig. B-3. Normalized Results from the Five-Cycle Cold-start Experiment (Ref. B-4)

The specific factors affecting exhaust emissions variability are listed under the corresponding source category in Fig. 2-11b. Results from one study (Ref. B-1, p. 6) that examined variability in CVS-CH exhaust emission tests of a given vehicle in a single test cell at several laboratories in repeated testing showed considerably more variability in a 1971 production car vs a 1975 catalyst-equipped, experimental control system and newer small car. Further data has led to the expectation that exhaust emission measurements specified in terms of the standard deviation as a percent of the mean for a 1975-76 vehicle in a given cell or from a cell-to-cell at one laboratory for

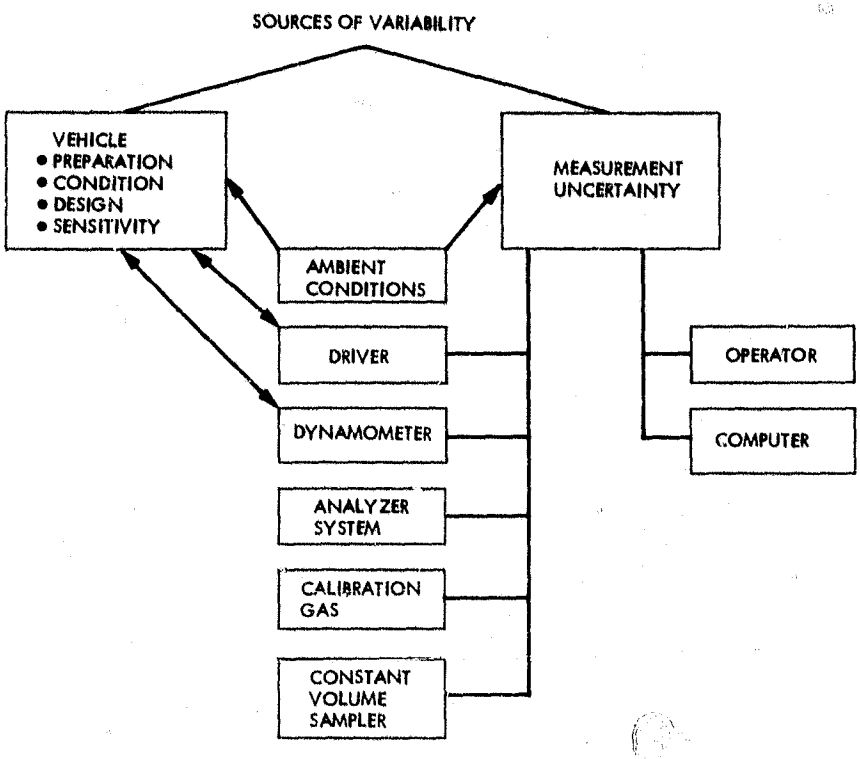


Fig. B-4. Sources of Variability (Ref. B-5)

most engine control system combinations may be in the following ranges:

HC = 10-25%
CO = 15-30
NO_x = 5-15

Many other tests including round robin emission tests have been carried out in the U. S. and Germany. Statistical analysis of these results indicated that some laboratories were suffering from systematic errors as high as 30% (Ref. B-1). The variability in tests of mass exhaust emissions is cause for major concern in testing to show that a given car complies with the emission standards. The emissions variability is influenced by the factors listed in Fig. B-5 (Ref. B-5). The vehicle type, driver and ambient conditions have significant influence on both HC and CO emissions. NO_x is also influenced by ambient conditions: however it is more strongly influenced by the vehicle loading.

Following the 1975 Federal Test Procedure, tests were conducted on vehicles whose emission levels were at or near standards of 0.41 g/mi. HC, 3.4 g/mi. CO, and 2.0 g/mi. NO_x.

The test results showed that the net variability caused by both measurement error and within-vehicle variability was ±19% of the mean of HC, ±33% of the mean for CO, and ±9% of the mean for NO_x as shown in Fig. B-6. These percentages are for production cars. Note that variability is defined as ±1.96 times the standard deviation divided by the sample mean ($\sigma/\bar{X}$). The estimates of variability include measurement error on within-vehicle variability for a 6-car sample and 8 tests per car.

According to Ref. B-5, "...The system deterioration estimated in this manner includes the effects of test variability at each mileage interval. Knowing the average levels of variability for all emission contituents, Monte Carlo simulation of a 10-car sample has shown that a range of deterioration factors (for example, 0.699 to 1.687 for CO) can exist due to variability alone when there is no true system deterioration." Thus, variability in tests could mask true deterioration. However, a linear least squares approximation of the data

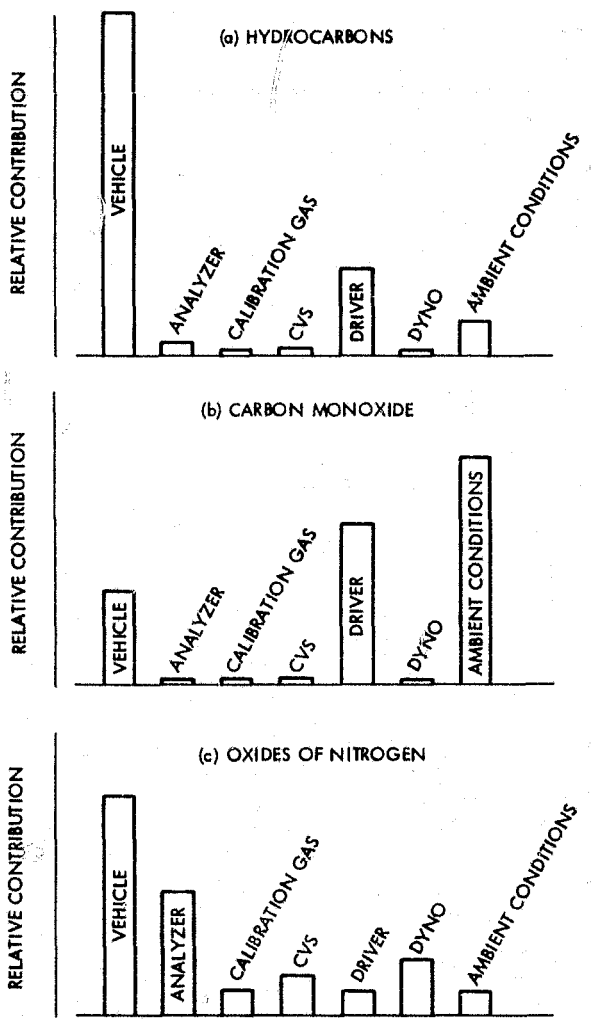


Fig. B-5. Sources of Variability-Relative Contributions (Ref. B-5)

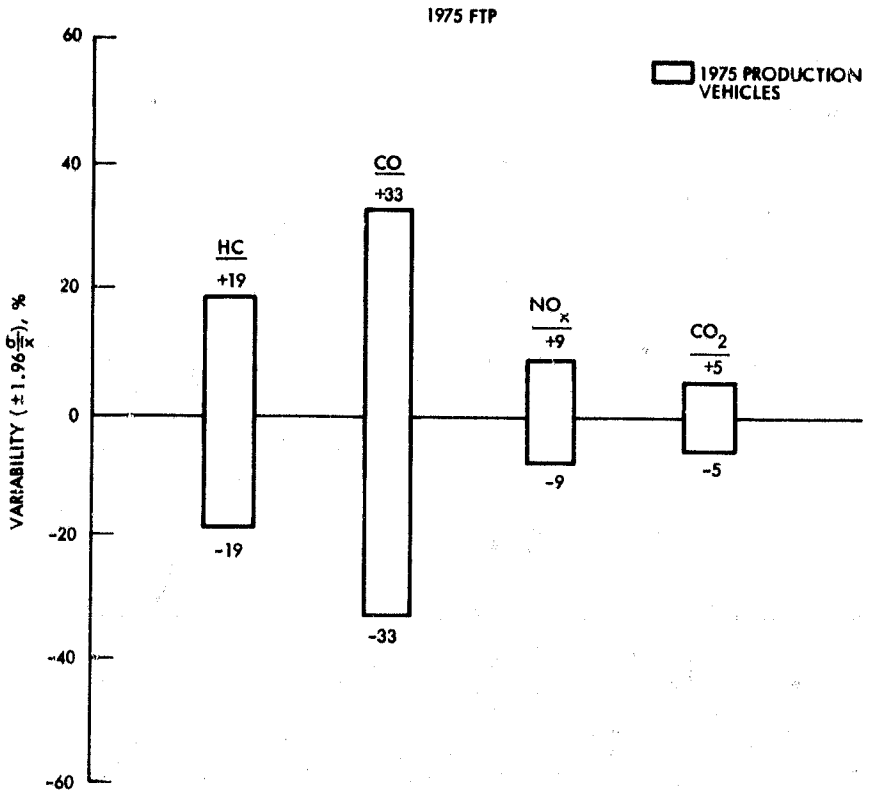


Fig. B-6. Range of Total Variability (Measurement Error + Vehicle Variability) (Ref. B-5)

for a 10-car sample showed a deterioration factor of 1.0, indicating in this case that for the same car retested the average emission may be significantly different from any single car emission measurement. Fig. B-7 (Ref. B-5) displays this observation and further supports the hypothesis that single measurements of cars in surveillance tests may not correspondingly produce a representative data point for determining a least squares estimate for predictions and for planning strategies. The use of a single measurement per car may be part of the cause of the data scatter and low correlation coefficients, which will be discussed later.

The significance of these test variabilities is shown in Fig. B-8a, b, c, where the relative influence of sources of variability are shown for HC, CO and NO_x emissions. In particular vehicle variability for HC and NO_x are dominating, while ambient conditions and driver variability dominate for CO. Each of these sources of variability (for example, CO) leads to as much as a ±33% total variability, which is a range for which 95% of the test measurements would fall for a single car if there was no variation in the true level of the vehicle. The inherent variability is further demonstrated again by results in

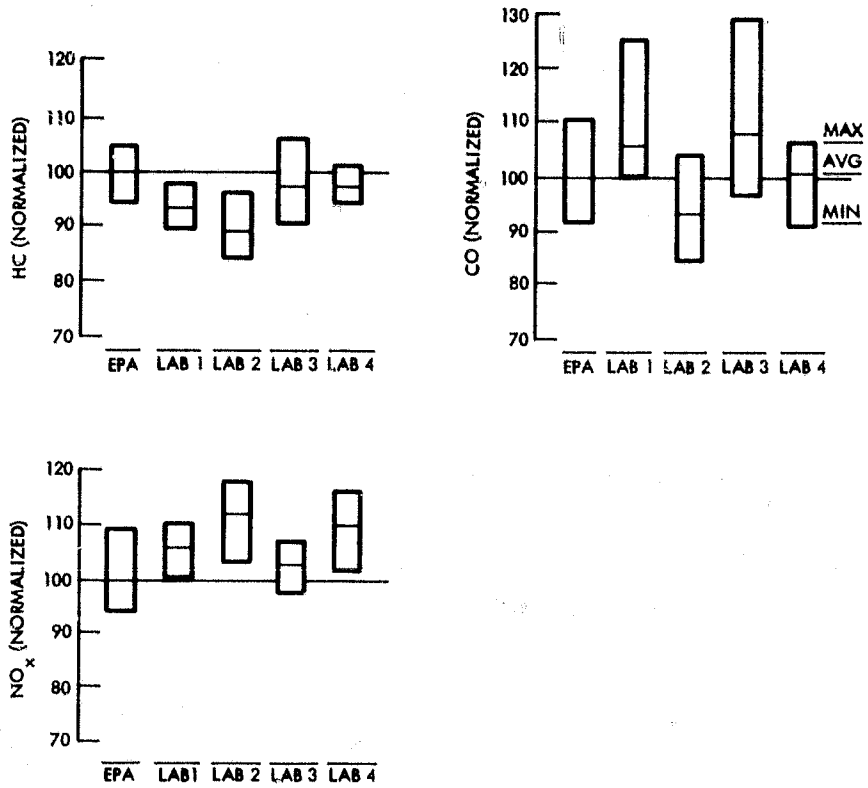


Fig. B-7. Laboratory-Laboratory Correlation Hot Start Tests (Ref. B-5)

Fig. B-9a, b, c, d, taken from a treatise on variability in Ref. B-5, thus making the assumption that the sample of one emission measurement (as is commonly made in surveillance tests) represents the mean for that car may be grossly misleading. As an exercise to demonstrate this concern the test variability data from Ref. B-5 for the six cars, measured eight times each for three different mileages under different site and driver conditions, is used as shown in Fig. 2-16. The implication in this analysis is that the fitted data could have almost any slope as indicated with the individual test results shown as plotted. Note that the means of the total samples show little change and characteristically show slight increase in the emission rates with mileage, even for these low mileage vehicles.

B.3 SURVEILLANCE TESTING

Surveillance testing to determine the exhaust emissions from in-use vehicles has been conducted by EPA for a number of years and CARB over the past four years. The EPA studies, known as the Emission Factor Program, were designed to gather emissions information to

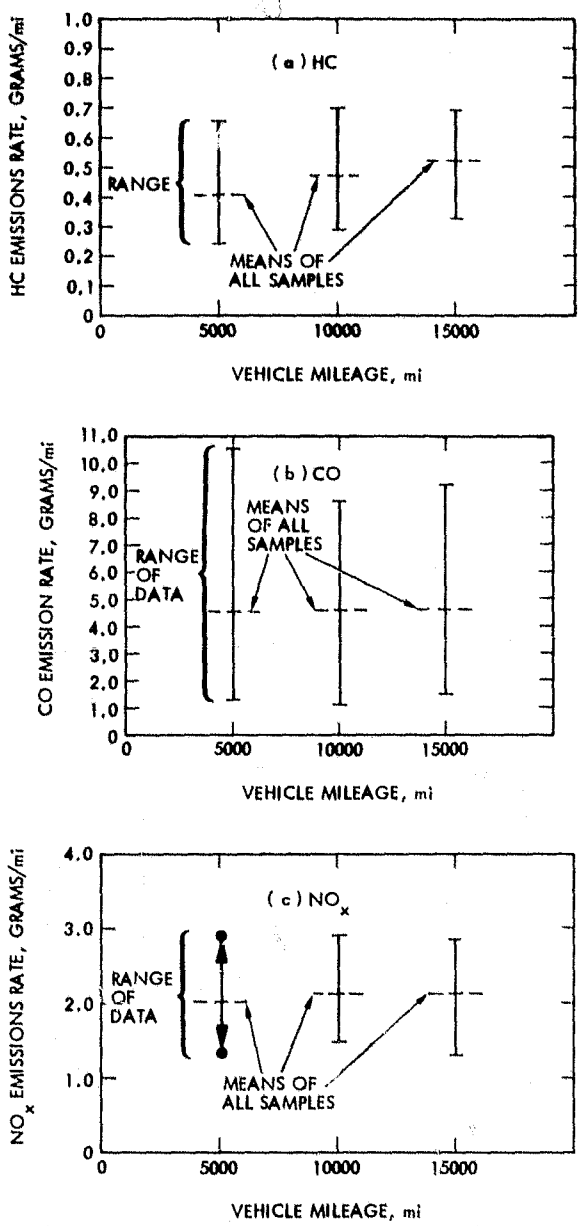


Fig. B-8. Test Variability in Exhaust Emissions from 6 Cars With 8 Measurements per Car After 3 Different Mileages (After Ref. B-5)

plan strategies for transportation management, to develop background for new regulations and to provide needed input for calculating and forecasting ambient pollution levels. The CARB program has the same goals except that it is directed toward California-peculiar problems and more stringent requirements both on new car standards and emission

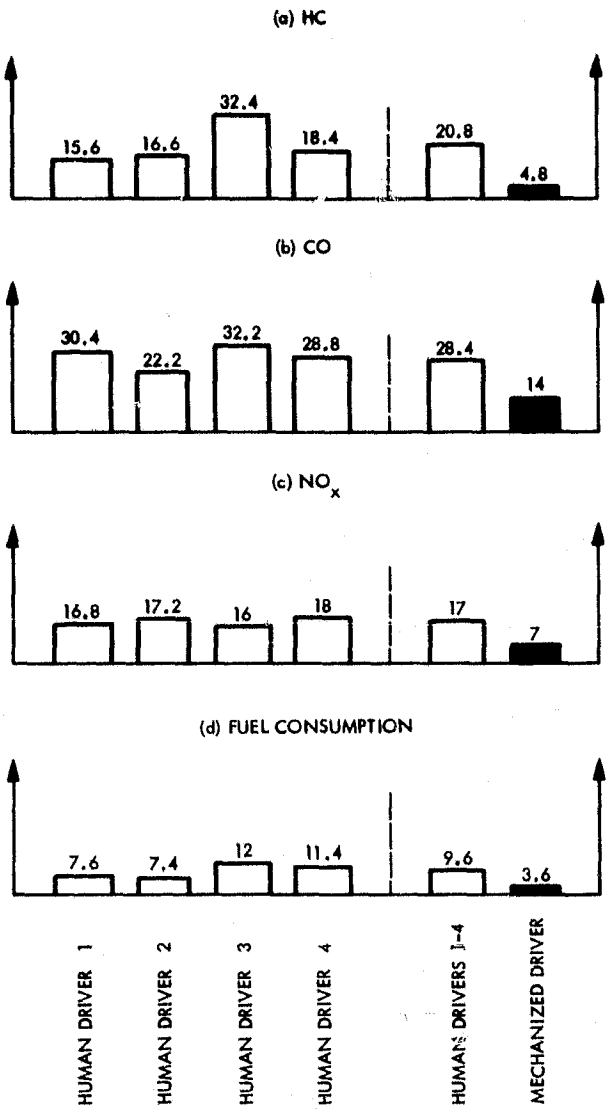


Fig. B-9. Comparison of Driver Variability During a Series of U.S. FTP 1975 Cold Start Tests (Ref. B-5)

control systems. The surveillance test results are primarily influencing strategies for transportation management to meet current and future standards. The data from these tests have a significant bearing on planning and demonstrating the need for an effective restorative maintenance plan for the U.S., and especially for California as mandated in the CAAA 1977 because I/M will be imposed where other measures do not lead to satisfying emission standards and regulations on the ambient pollutions.

The Restorative Maintenance Evaluation Program that EPA started in 1976 is examining the poor emission performance of in-use vehicles by investigating and quantifying individual and combined effects of defects and disablements or maladjustments on exhaust emissions and fuel economy. These results are specifically being used to (1) assess the effectiveness of the LDV Certification Process in relationship to the performance of in-use vehicles, (2) improve requirements for engines and emission control systems and (3) build information for planning I/M programs (Ref. 2-5).

A timely discussion on the further development of the data base to update the emission factors for catalyst-equipped California cars that were estimated by EPA (Ref. B-6) in March 1978 was reported by CARB in Ref. B-7. The results of the CARB tests are based on in-use passenger cars rather than relying on certification or assembly line data. The EPA results, which are also used for comparison purposes, were based on tests of fleets of vehicles in customer service and upon prototype testing. Both sets of test data evaluate the emissions performance of in-use passenger cars, but it should be pointed out that the small amounts of in-use data leading to using averages of the test samples to data do not demonstrate significant correlation. A typical characterization of the scatter of a small sample of data is demonstrated in Fig. B-10.

The primary reason for comparing the earlier EPA emission factors with the more recent CARB emission data is to show:

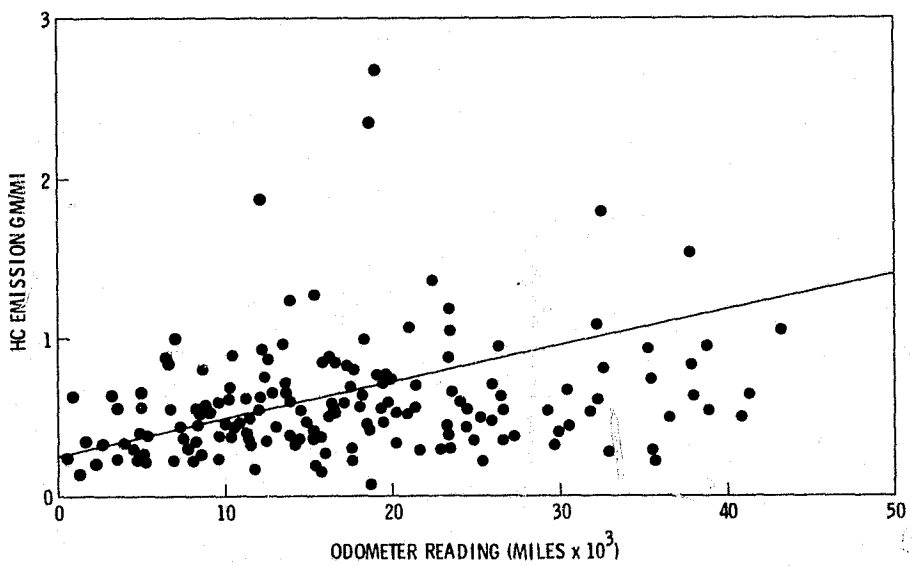


Fig. B-10. Scatter Diagram and Regression Function for Hydrocarbon Emissions from 1977-78 Cars

(1) The effect of updating the data base with results from a relatively extensive testing of in-use California vehicles that were specifically designed to meet emission standards of 0.9 HC/9.0 CO/2.0 NO_x in 1975 and 1976 and 0.41 HC/9.0 CO/1.5 NO_x in 1977 through 1979. Note that the Federal standards are less stringent than the California standards for 1975 through 1979 (see Table 2-1).

(2) The potential concern with in-use vehicles being able to meet the standards after about 25,000 miles, even though significant improvements have been demonstrated with the use of the new emission control systems.

(3) The significant potential emission reductions through the use of I/M which is manifested by the difference between "as received" and "repaired" vehicle test data. The details of differences between "as received" and "repaired" are fully discussed with qualifications in Ref. B-7.

(4) The significant improvements in the 1977-1979 cars over the 1975-1976 cars manifested in both the EPA and CARB cars, indicating that emissions are decidedly less with newer model.

Figures B-11 through B-16 show comparisons of the NO_x, HC and CO emissions based on EPA and CARB data for 1975-1976 and 1977-1979 cars as a function of odometer readings of the samples. Two features that are important to modeling and trend forecasting efforts are clearly shown in Figs. B-11 and B-12. In Fig. B-11, the EPA data indicates that the California cars would have met the California NO_x standard up to about 30,000 miles while the CARB data shows that the average car will not meet the standard "as received" after being in-use, but the deterioration is not significant even out to 70,000 miles if the cars are repaired based on evaluation of idle test results and standards established for the California I/M program.

The contrast between the EPA and CARB 1977-1979 vehicle test data is more strongly shown in Fig. B-12. In this figure the EPA data shows significant deterioration after meeting the standard up to about 25,000 miles, while the CARB data shows that the average car is not deteriorating relative to meeting the NO_x standard. It further shows that the average repaired California car meets the California NO_x standard.

The HC emissions test data in Fig. B-13 shows that after 20,000 miles the 1975-1976 CARB cars typically do not meet the California emissions standard even after repair. Fig. B-14 shows that the 1977-1979 "as received" cars exceed the standard after approximately 8,000 miles. The HC emissions standard for the 1977-1979 cars is about 1/2 that for the 1975-1976 cars. Fig. B-15 shows that HC emissions

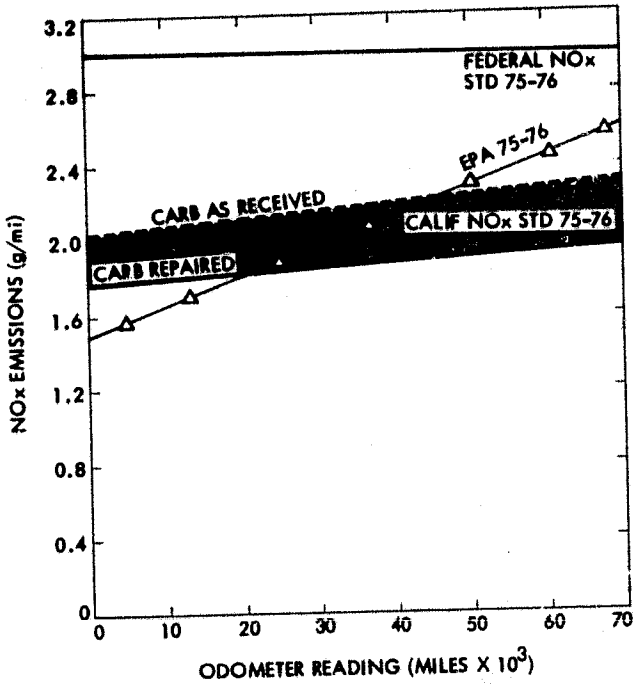


Fig. B-11. Comparison of NO_x Emissions versus Odometer Reading Using CARB and EPA 1975-76 Cars Surveillance Test Data

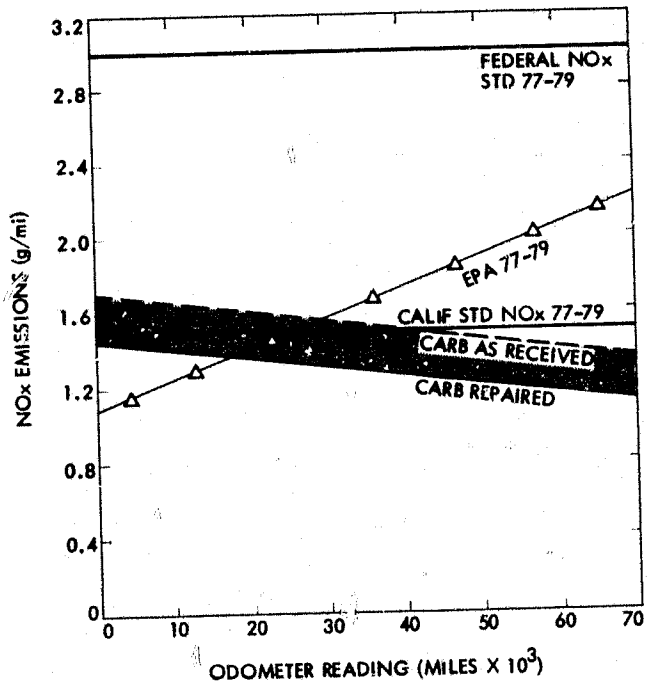


Fig. B-12. Comparison of NO_x Emissions versus Odometer Reading Using CARB and EPA 1977-79 Cars Surveillance Test Data

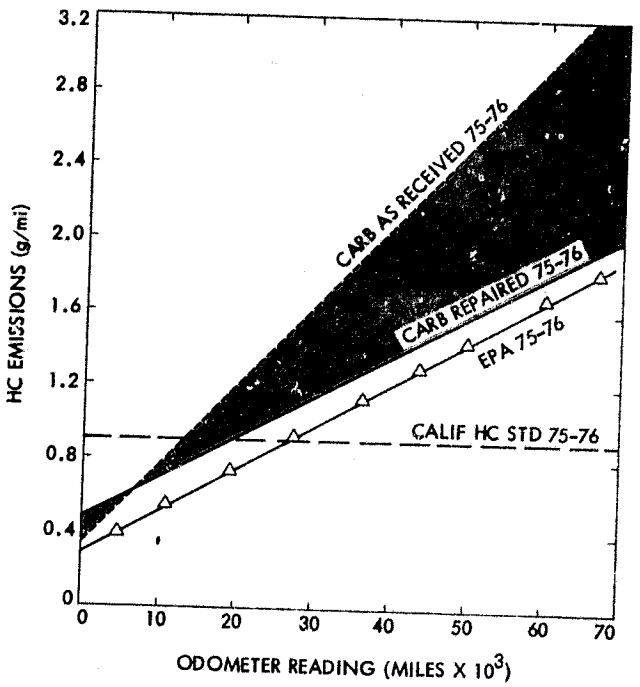


Fig. B-13. Comparison of HC Emissions versus Odometer Reading Using CARB and EPA 1975-76 Cars Surveillance Test Data

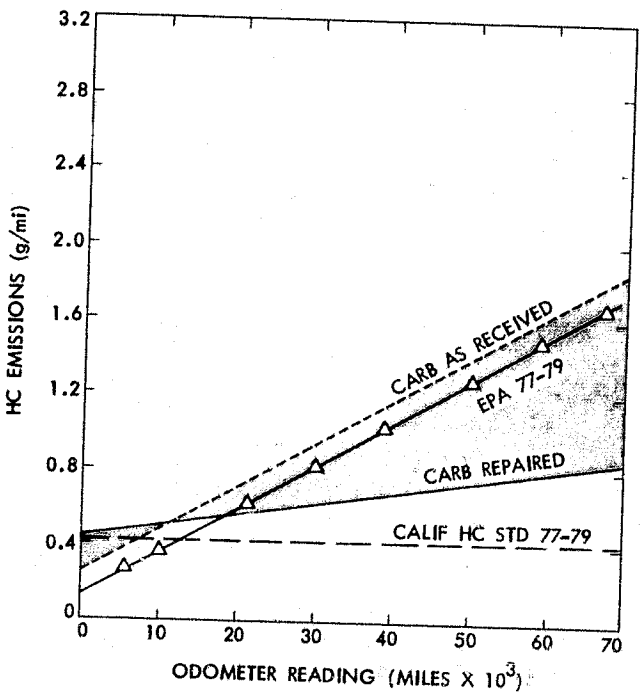


Fig. B-14. Comparison of HC Emissions versus Odometer Reading Using CARB and EPA 1977-79 Cars Surveillance Test Data

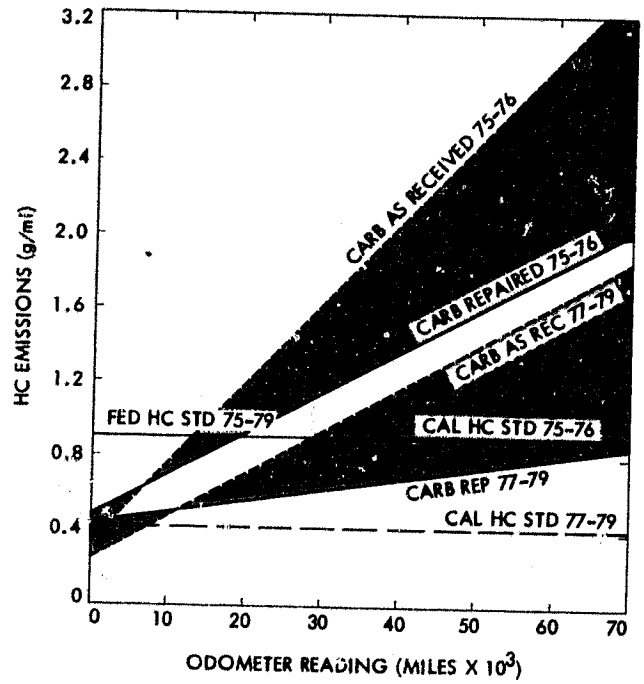


Fig. B-15. Comparison of HC Emissions versus Odometer Reading Using CARB and EPA 1975-79 Cars Surveillance Test Data

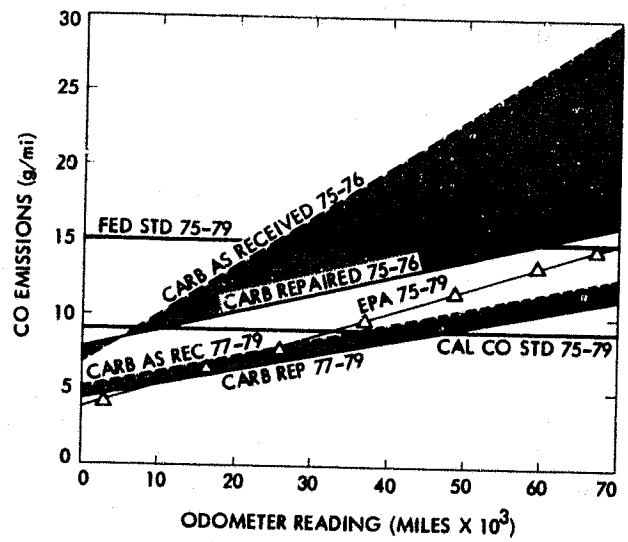


Fig. B-16. Comparison of CO Emissions versus Odometer Reading using CARB and EPA 1975-79 Cars Surveillance Test Data

from the 1977-1979 cars are considerably less than those from the 1975-1976 car It is important to note that the repaired 1977-1979 cars meet the 1975-1976 California HC standard even though they do not meet the 1977-1979 standard. Similar comments can be made for CO emissions. Fig. B-16 shows that the CO emissions for both the repaired CARB 1976-1976 cars and the "as received" 1977-1979 cars meet the Federal CO standard. It also shows that the EPA 1975-1979 emission factor deterioration curve fits the 1977-1979 data better than the 1975-1976 data. Since both the Federal and the California CO standards have remained constant from 1975 to 1979 (Fig. B-16), the improvement in the 1977-1979 cars could be due to improvements in technology and manufacturing processes. The CARB "as-received" 1975-1976 cars do not meet the Federal standards after approximately 25,000 miles. If these 1975-1976 cars are repaired through the CARB I/M program, the average car meets the federal standard until 60,000 miles. At that point it would exceed the California standard by about 65%.

About 67% of the vehicles in-use now have odometer readings of more than 50,000 miles; vehicle miles traveled by these vehicles is 57%. These vehicles, however, contribute more than 74% of the LDV CO emissions and 74% of the HC emitted (Ref. B-8.) Therefore, without repair the emissions from the total car population would be much larger.

The influence of repair on HC emissions as well as the improvement in emission control systems is dramatically displayed in Fig. B-17 by the regression curves for the 1975-1976 cars vs 1977-1979 cars. Although California HC standards will not be met by the average in-use vehicle, an effective I/M program could potentially reduce HC emissions by almost a factor of 2 for the average car having 50,000 miles on the odometer.

Along with this potential reduction in emissions expected with I/M, the influence of regular inspection could lead to improvements in vehicle driving habits, effective use of emissions control mechanisms and vehicle function. Specific factors such as these related to in-use vehicles are discussed below to highlight their variability.

Variability of In-Use Vehicles and Uncertainty in Applying Average Emission Factors. Many factors such as driver operation, emission control system, and traffic action and controls interact during the operation of a given vehicle in urban and suburban traffic to cause large variations in exhaust and fuel consumption. Some of the more realistic variabilities lie in actual on-road use of the vehicles. Some examples of how these factors influence the fuel consumption and exhaust emissions of a vehicle in urban traffic are displayed to examine the practicability and reliability of these types of quantitative data bases. In particular (Ref. B-9), a recent investigation by the Traffic Science Department of General Motors Research Laboratories has resulted in rather consistent descriptions of how fuel consumption and hydrocarbon emission depend on traffic conditions. The general finding was that urban traffic conditions may be quantified by a single characteristic--the average speed of the traffic. In this case the study indicates that HC emissions may be modelled as a simple function of trip

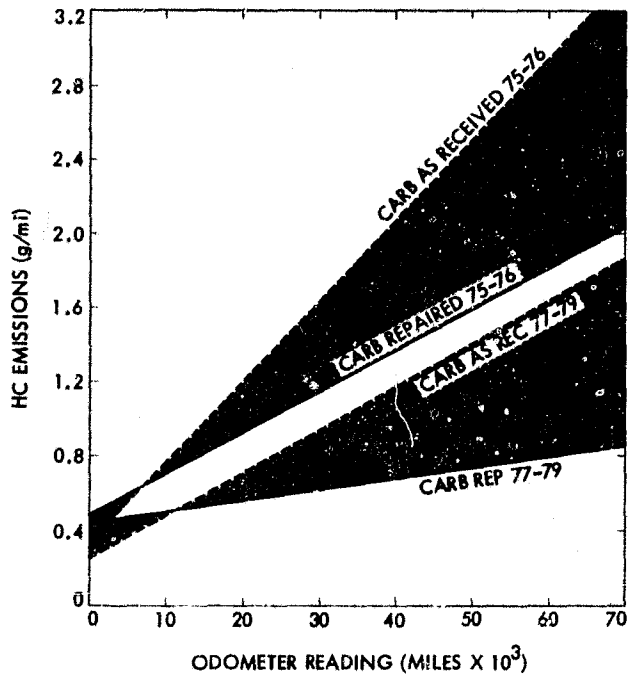


Fig. B-17. Comparison of HC Emissions versus Odometer Reading Using CARB 1975-79 "As Received" and Repaired Cars Surveillance Test Data

speed based on analyses of small segments of FTP data. It is important to note that in this study no simple analytic models were found for CO and NO_x.

Considering this latter conclusion, there is still a need for a way to estimate consistency of various measurements and its influences on mobile sources emissions forecasts and transportation strategies.

In the study of emissions, all 12 cars were equipped with catalytic converters and emissions both before (engine-out) and after (tailpipe) were studied. Because exhaust emissions do not depend simply on, for example, vehicle mass, as is approximately so for fuel consumption, a relative relationship between exhaust emission and traffic characteristics was analytically correlated for HC only. An important note of this study was that engine-out hydrocarbons, being larger and not subject to additional statistical variability associated with passing through the catalyst, yielded more consistent car-to-car relations than the tailpipe emissions (Ref. B-9). Fig. B-18 shows the results of relating engine-out HC to average trip time per unit distance $\bar{t}$, based on the 12 cars run on the FTP (see Ref. B-9 for details). This relationship indicates that the variable $\bar{t}$ explained more than half of the variance in HC for 11 of the 12 cars and the average variance explained by $\bar{t}$ was more than 70% for these 11 cars.

The intent of this examination of a single variable is to compare it with the more complex equations derived by EPA, which also indicate the variability with this single variable. This feature is also described in Ref. B-9 and illustrated by Fig. B-19. In this figure the ratio of HC emissions at different speeds is shown for the 11 cars with the 70% correlation. It also shows EPA data that was used to derive the ratios used in EPA Document AP-42 (Ref. B-10) to derive average speed exhaust emissions. In the updated version of AP-42, the draft of Mobile Source Emission Factors (Ref. B-6), composite exhaust emission factors are specified to be calculated per pollutant and model year. The equation may be evaluated for average speed, ambient temperature, humidity, fraction of cold operation and fraction of hot start. It includes the fraction of annual travel and special load conditions (such as air conditioning and trailer towing). The variability in the single variable, average trip time per unit distance previously discussed and demonstrated in Figs. B-18 and B-19, may be indicative of the magnitude of variabilities introduced in the other variables cited. Taken as a composite set of variables and the use of averages per model, year, etc., this averaging might compensate for uncertainty about the average, but the potential for error in projecting should be specified so that as the data base becomes more certain the confidence in projections increases.

As of now the uncertainty on the average emission factors may be as much as the factors of 2 to 3 that are seen on the variables in the data base. The question is that since the uncertainties are being narrowed as more test data becomes available and as the better emission control technologies are applied, will the averages adequately predict the future emissions or overpredict it? A sensitivity analysis

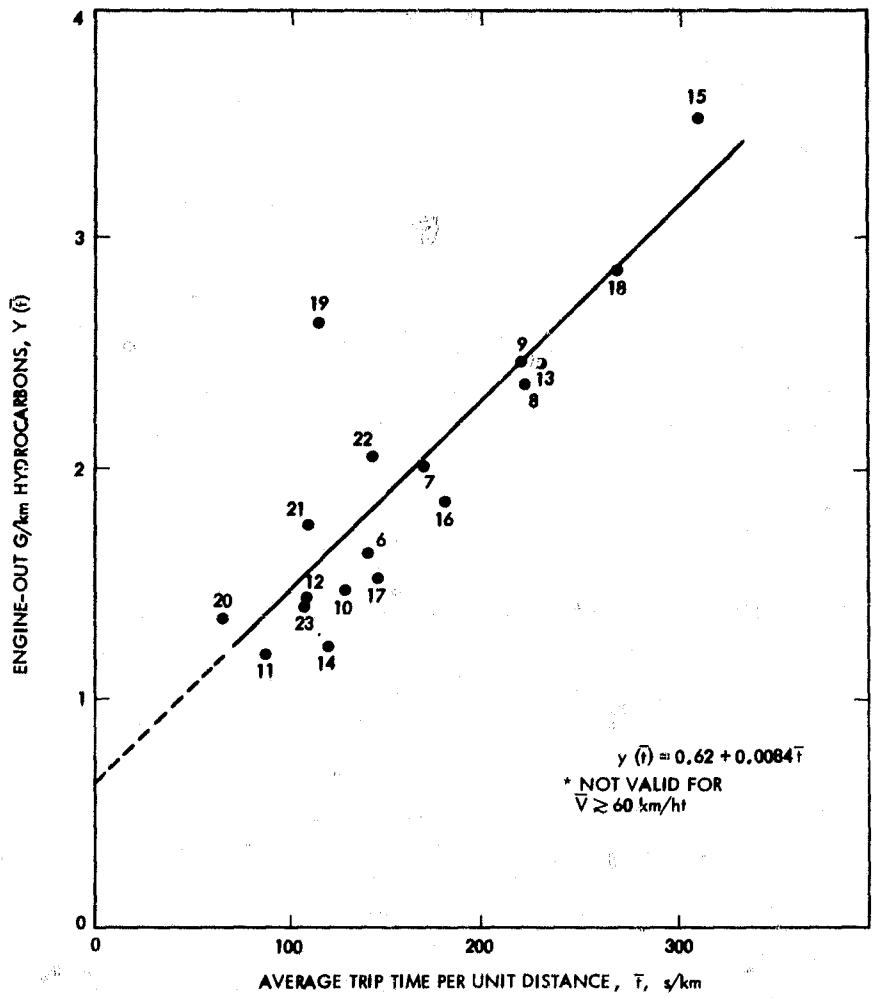


Fig. B-18. Engine-Out HC Emissions versus Average Trip Time per Unit Distance (Ref. B-9)

year by year from 1970 through 1982 should be performed practically with available data to examine each variable in the emissions factors equations.

Effect of Ambient Conditions on Emissions. The ambient temperature differs considerably from the standard test conditions of 20 to 30°C (68 to 86°F) during many of the tests conducted to predict emissions for the total population of motor vehicles and for the current and future years. In addition, the behavior of motor vehicles in ambient temperature depends upon design and type of emission control system. Because of the dominance of the new control technology in late model cars and the fact that well over 80% (Ref. B-11) of the cars in the U.S. have emission control devices and approximately ∿75% of the ≥1975 models have catalytic converters (100% in California since 1975),

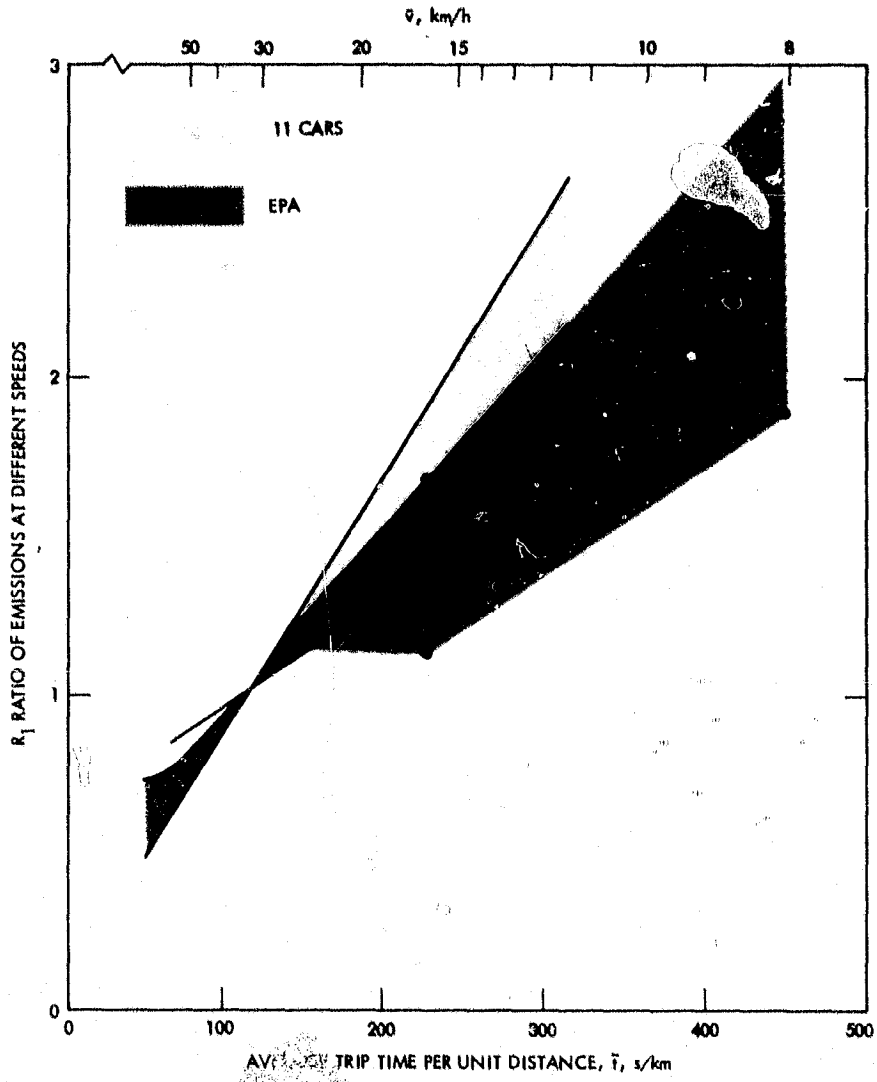


Fig. B-19. Hydrocarbon Emissions versus Average Trip Time per Unit Distance

the effect of temperature on catalytic converter cars is discussed to highlight its importance to transportation planning strategies and its influence on emissions forecast based on test data.

In Ref. B-12 regulated pollutants were higher at -30°C than at 20°C (standard FTP test conditions) by the following factors:

HC - 3.5 to 9.2 higher
CO - 2.4 to 6.4 higher
NO_x - 1.1 to 1.4 higher

It is important to note that the phase most temperature-sensitive for HC and CO was cold start; but for NO_x the effect was fairly constant but slight throughout the three phases of the Federal Test Procedure. For comparison purposes and to demonstrate the range of the effect of ambient temperature, California cars are compared with U.S. 49 states cars (all U.S. manufactured) in Fig. B-20a, b, c. The cars compared are from 1975 model year with exhaust gas recirculation, manifold air injection and oxidizing catalytic converters. Particular note was made of the catalytic cars with air injection because of the significantly lower level in emissions compared with those without air injection. This was explained by the lack of oxygen to oxidize the CO and HC present under rich choke conditions. The point made was that the needed oxygen during cold start would maximize the efficiency of an oxidizing catalyst, and air injection would generally be beneficial for internal combustion engines to achieve the most efficient emission control at lower temperatures. The design of emission control systems for new cars since 1975 has predominantly included air injection or air pumps.

A reasonable comparison for California is that at $0^{o}C$ ($32^{o}F$), a common early morning winter temperature for several months throughout the year, the HC and CO emission were about 50% higher during the cold start phase while the ambient effect was negligible for the other test phases.

Therefore, cold ambient temperatures below $-10^{o}C$ would considerably increase the HC and CO emissions but not the NO_x during the cold start phase. This effect of temperature is pronounced in cars without air injection. After warmup the ambient temperature shows a large effect. The largest effect of ambient temperature on CO emissions is shown in Fig. B-21 for the cold transient phase. Fig. B-22 also demonstrates the effect of ambient temperature during each test phase of the FTP 1975 and highway driving. Ambient temperature causes variations as much as a factor of 3 during hot transient and a factor of 5 during highway driving.

Soak Time Effects on Car Emissions. Soak time effects based on a 5-car sample have been determined following the 1975 Federal Test Procedure for light-duty vehicles and using a Clayton Model ECE-50 (Ref. B-13). By using a 16-hr soak period, which is fairly representative of an overnight soak as the baseline, the results were discussed as a deviation from the baseline as shown below:

$$\% \text{ of baseline} = \frac{\text{emission rate after given soak}}{\text{emission rate after 16-hr soak}} \times 100\%$$

Emission rates and fuel consumption for each vehicle following specific length soak periods are shown in Fig. B-23. Normalized results for the 23-min urban dynamometer driving schedule (UDDS) are plotted vs $\log_{10}$ of soak period length. The log-normal presentation of data was found to adequately define the changes early in the soak period as well as those after very long soak periods. The HC emission rates followed essentially the same trend for all vehicles, increasing with soak period length over the range of soak periods tested. The CO emission rates

were stable for four of the five cars following 2-hour and shorter soak periods and increased with increases in soak period length thereafter. The type of exhaust after treatment had a discernible effect on the level of CO following short soak periods. The CO emission rates following 4-hr and longer soak periods increased with increases in soak period length for all cars.

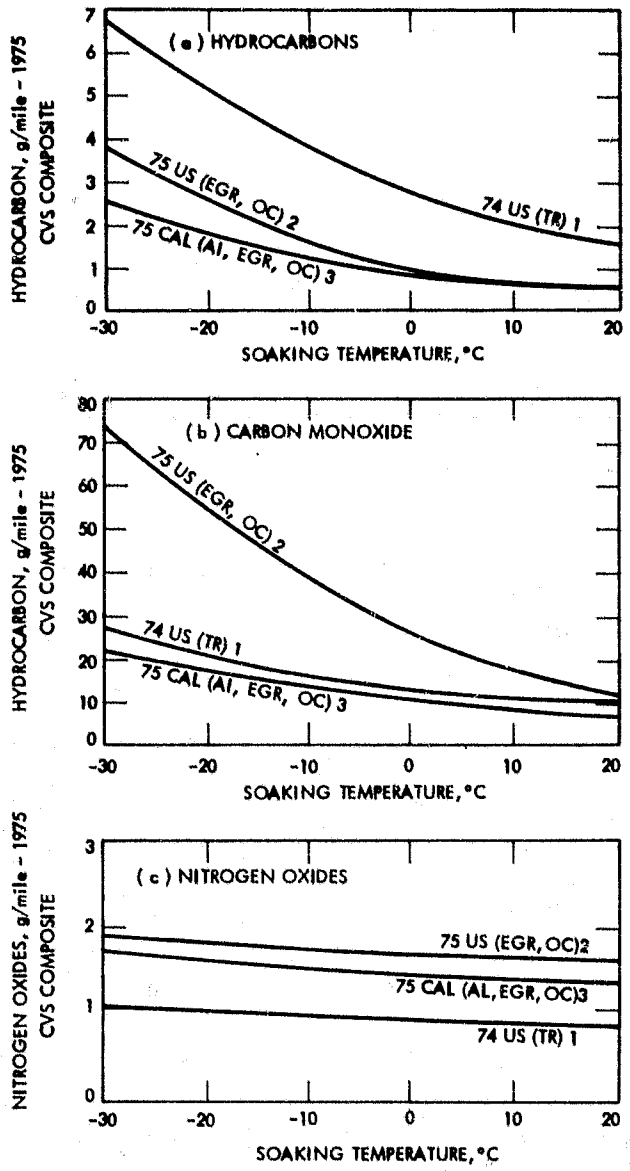


Fig. B-20. Influence of Soaking Temperature on Emissions (Averaged for Vehicle Groups) (Ref. 2-16)

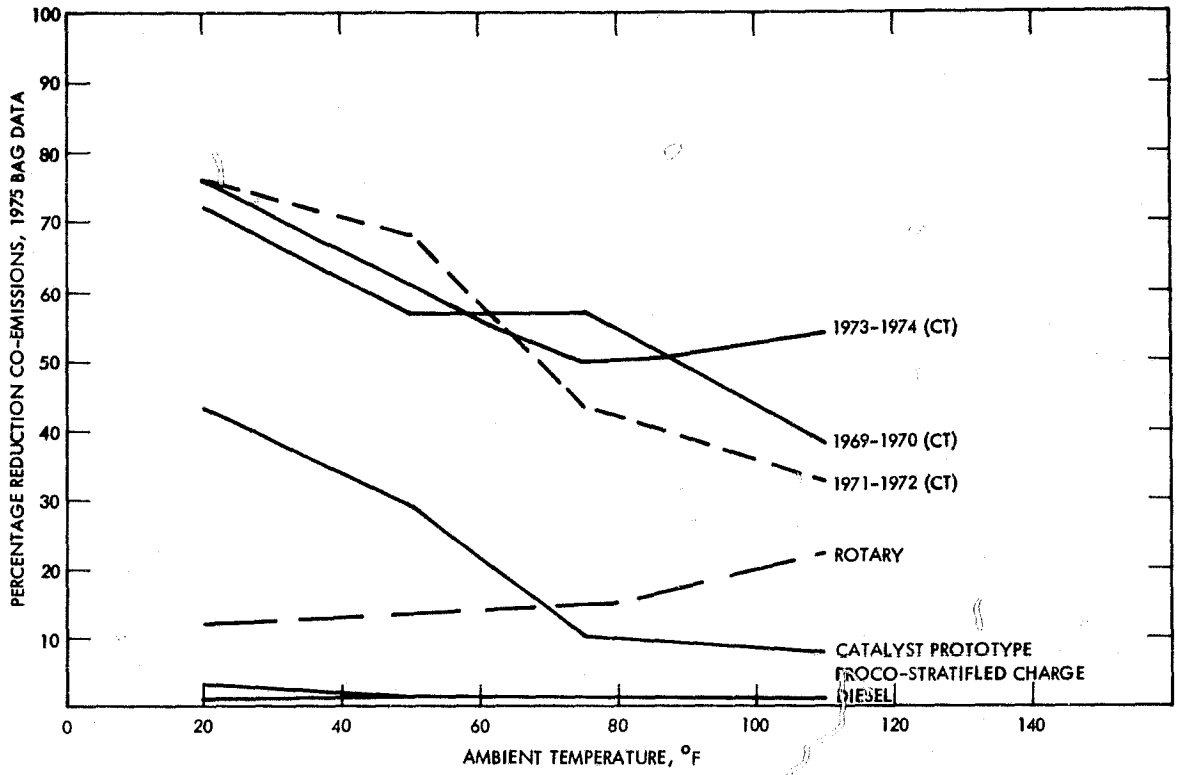


Fig. B-21. Percentage Reduction in CO Emissions versus Ambient Temperature

The NO_x emission rates for the 23-min cycle increased with increases in soak period length over the entire range of soak periods tested; NO_x following a 10-min soak period was generally 80 to 90% of the baseline soak period value. Composite fuel consumption for the UDDS is the most predictable of the items measured. For each of the five vehicles tested, the trend in fuel consumption is a remarkably linear function of the $\log_{10}$ of the soak length.

A characteristic common to the emission rates for the majority of the vehicles tested is higher levels following a 36-hr soak period than following the 16-hr soak. Except for NO_x, levels following an 8-hr soak period were lower than the 16-hr soak levels. This indicates that the shortest soak period allowable by the FTP would yield the lowest emission rates in most cases.

The behavior in the transient portion of the cycle is considered qualitatively identical to the composite cycle results. The HC emission rates were essentially constant in this portion of the cycle regardless of soak period length. The important results are that the behavior during the transient phase of the cycle is qualitatively identical to the composite of the transient and stabilized phases. This means that the characteristic behavior vs soak time occurred

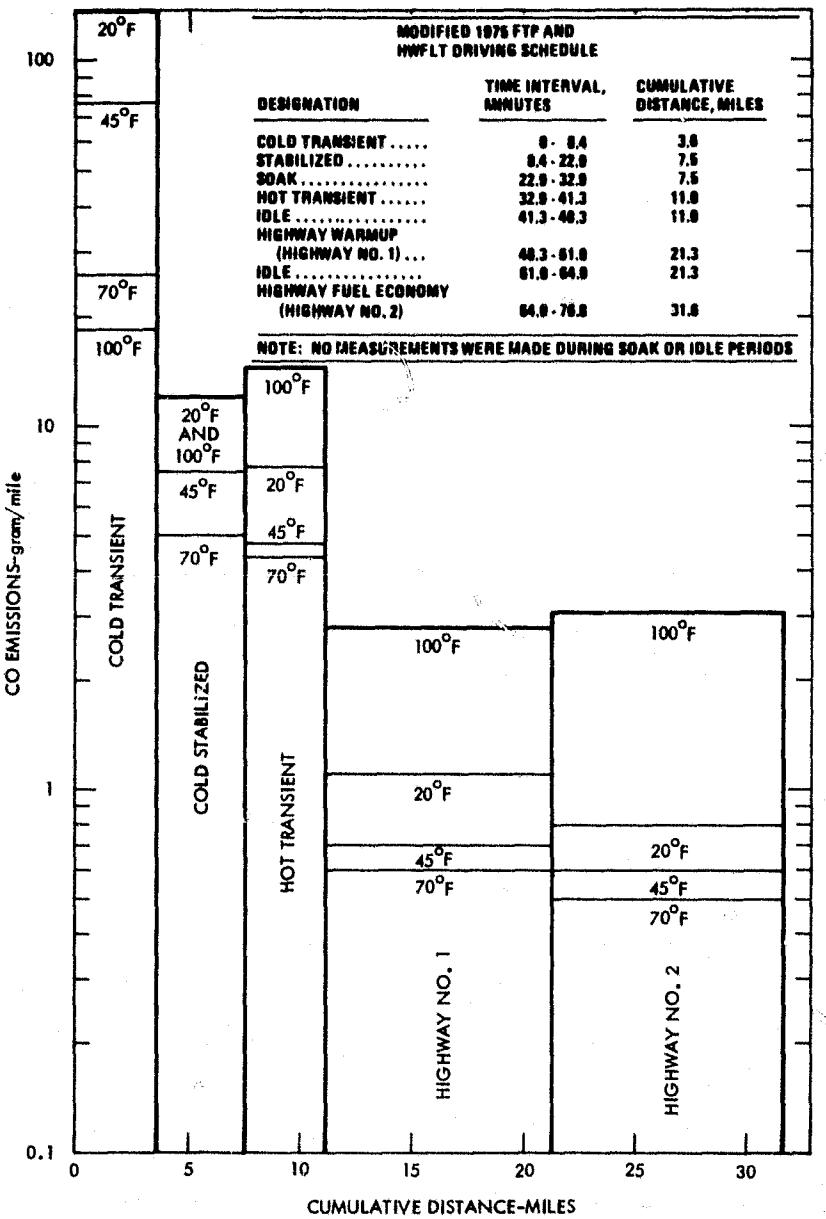


Fig. B-22. Range of Ambient Temperature Effects on CO Emissions Over the FTP 1975 and Highway Driving Test Phases

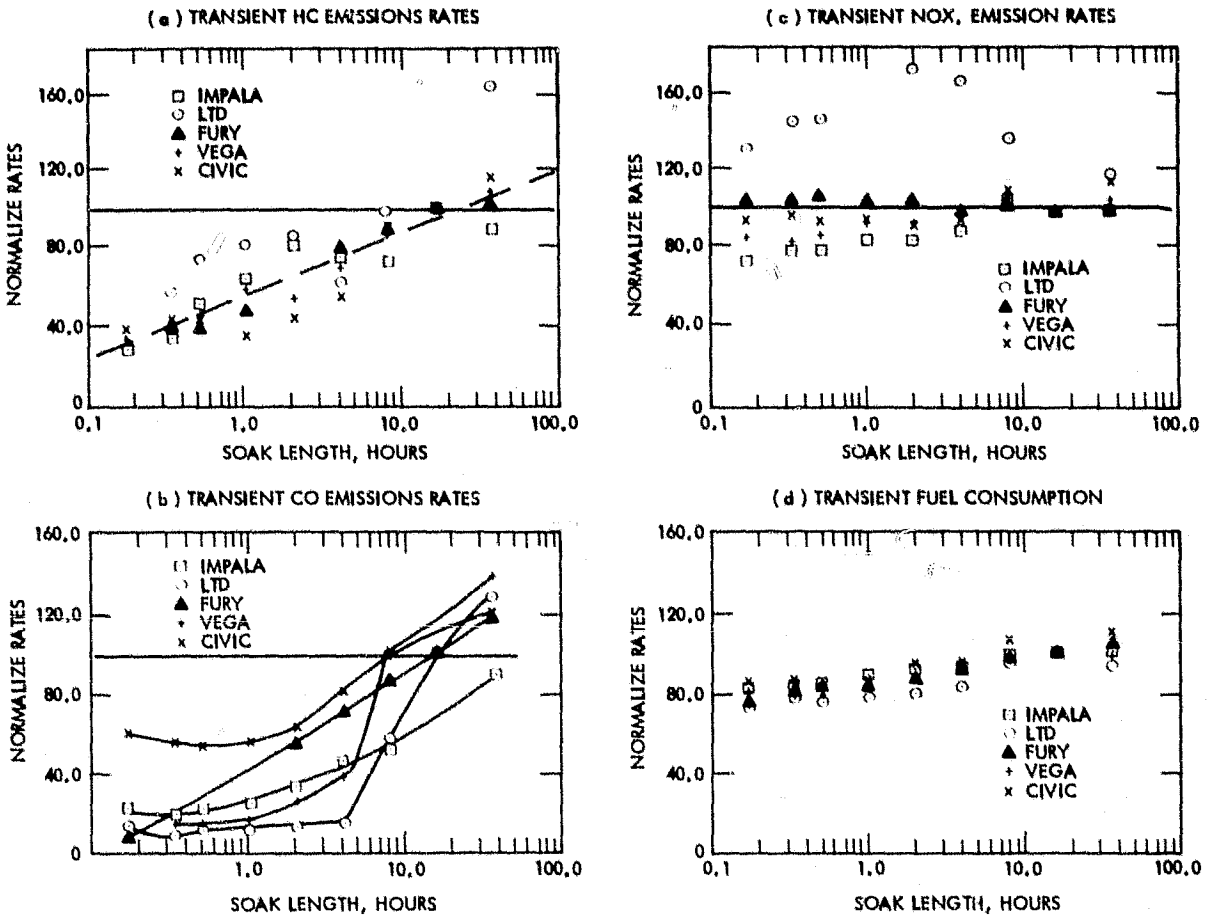


Fig. B-23. Normalized Transient Emission Rates and Fuel Consumption (Ref. B-13)

during the short (505) transient phase. The influence of increased soak time on the stabilized phase showed weak trends for CO and NO_x and the HC was essentially constant regardless of soak period length.

Briefly, the test procedure included driving the test vehicle over the 23-min, 7.5-mile-long EPA Urban Dynamometer Driving Schedule one time. The car was then soaked for a period of time at 24 + 2°C (75+3°F). The Federal Register permits the soak to be between 20 and 30°C (68 and 86°F). The vehicle was then driven over the UDDS cycle from a cold engine start. The first 505 seconds of the UDDS is referred to as the "cold transient" phase, and the remainder the "stabilized" phase. The emissions were measured following soak periods of 36, 16, 8, 4, 2, 1, 1/2, 1/3, and 1/6 hours. Emission of hydrocarbons (HC), carbon monoxide (CO), carbon dioxide (CO_2), and oxides of nitrogen (NO_x) were measured.

B.3.1 Idle Test Emissions

The importance of developing reliable information during the idle phase of the FTP is shown in Fig. B-24. Twenty-one percent of vehicle operation is attributed to cold start, 27% to hot start, and the remaining 52% is stabilized including 18% idle operation (Ref. B-6). The emissions during idle are at least comparable or in larger amounts for CO and HC during other phases of the driving cycle. However, the NO_x emission is typically minimum during idle compared with that at speeds above 20 mph.

The important point from Fig. B-24 is that the idle HC emissions may be a significant contribution to the total HC emission. For many of the older cars, 1974 and earlier, the emission factors are significantly larger than those for 1975 and new model cars. Again, it is worth noting that 74% of the HC emitted comes from cars that have greater than 50,000 miles but that these cars only cover 57% of the vehicle miles traveled (Ref. B-8).

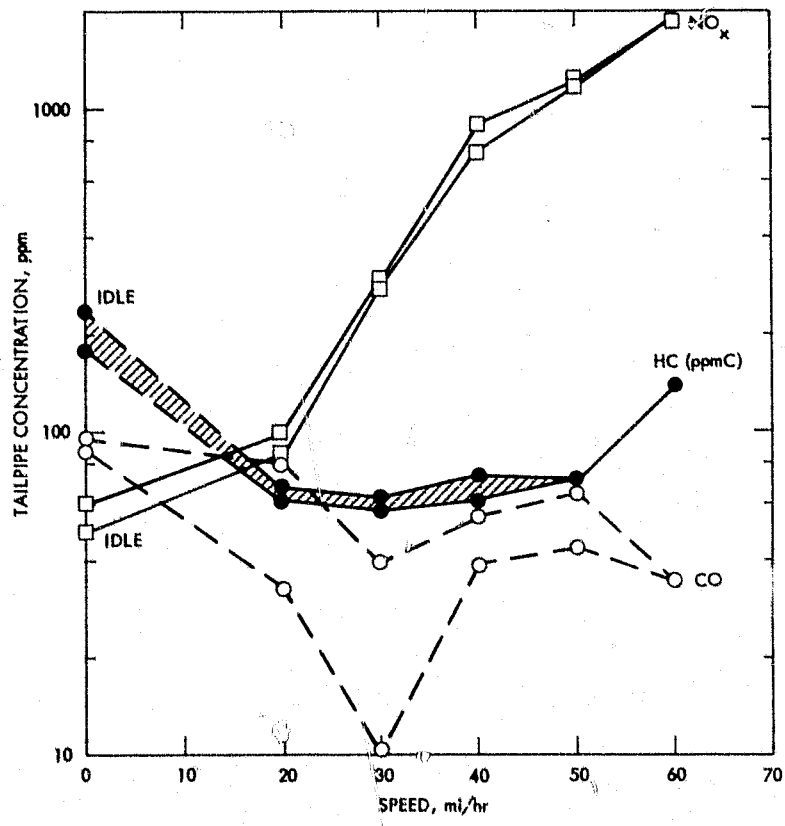


Fig. B-24. Hydrocarbon Emissions from Steady Speed Tests--1973 Model Equipped With 1975 Catalyst and Emission Control Devices (SAE 760036)

The concern for the large HC and CO emissions during idle is further supported by the idle emission results shown in Fig. B-25 for Los Angeles cars (data taken from the TY75 emissions factor program). This figure shows the mean values of HC and CO tests results plotted as a function of model years. In addition, the $\pm 1\sigma$ error estimate is shown to demonstrate variability of the data. The implication of the large variation is that the idle emissions from the non-repaired cars may be a significant contribution to the total HC emission.

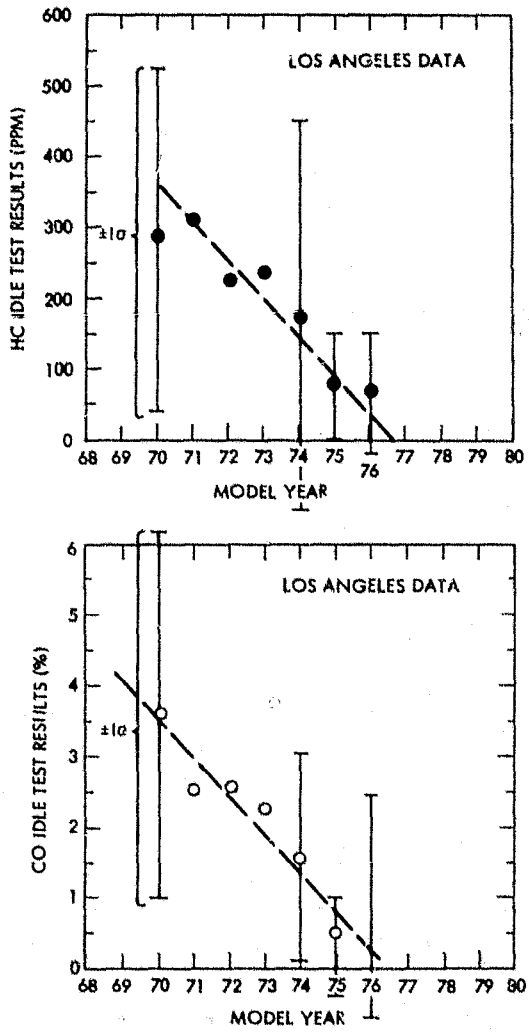


Fig. B-25. Idle Emission Results For Los Angeles (FY75 Emissions Factor Testing Program)

B.3.2 Evaporation Emissions

In the past, a lack of speciation data for passenger cars has been identified as a primary problem in establishing source inventories of individual hydrocarbon emissions. In addition, the rates and profile of HC emission vary significantly with fuel, driving patterns, ambient conditions and types of emissions control systems used on the vehicle. Currently used control technologies include:

(1) Crankcase emissions: positive crankcase ventilation technologies.

(2) Tailpipe emissions: catalytic converters technologies.

(3) Evaporation emissions: absorption--regenerative carbon canister.

Even though the catalytic converters reduce both the mass of HC and the photochemical reactivity of the HC mixture, the impact of evaporative control devices on HC emission has been demonstrated.

The venting of fuel vapors from the carburetor front chamber and vehicle fuel tank can constitute a sizable source of HC emissions in the absence of appropriately functioning control devices. Two types of evaporated losses are measured: Losses occurring over a 1-hr period while fuel (in the vehicle tank) was raised in temperature according to a prescribed schedule are termed diurnal soak losses. Losses occurring during a 1-hr period while the vehicle cools following the completion of a complete cold start 1975 FTP are termed hot soak losses. The measurement of emissions is made while the vehicles are enclosed in the Sealed Housing for Evaporative Determinations (SHED).

Summary results from the SHED tests are presented in Table B-1. A determination of the mean evaporative losses was made for 20 1976-model-year cars in both Denver and Los Angeles. Also included are the mean and standard deviations for combined losses, which are the results of calculations from each vehicle that are intended to represent the total average loss for a vehicle in a day. All three mean losses are significantly higher for Denver than for Los Angeles, as would be expected due to the altitude difference.

For some model years the evaporative HC emission may be nearly 1/2 the total mass of HC emitted from the vehicle during its use. To make this comparison, the Federal test procedure (Ref. B-14) gives an equation that converts evaporative emission (gram/test) to gram/mile using diurnal and hot soak mass emission factors (Ref. B-6) for these evaporation tests. The diurnal test simulates emissions experienced during fuel system heat build resulting from daily night to afternoon ambient temperature increases. The hot soak test simulates emissions experienced on parking the car after an urban driving period.

In a recent 4-car test series (Ref. 2-2) using a 1963 Chevrolet, a 1977 Mustang, a 1978 Monarch, and a 1979 LTD-II to perform a comparative study of evaporative and tailpipe emissions, the 1963 and 1977 car data suggested about 50% of the aggregate emission are evaporative. The 1978 and 1979 cars had 35 and 39% of their aggregate HC emissions from evaporation sources. These test results indicate that significant further reduction is necessary to get down to about 27% of the total aggregate HC emissions from evaporative sources. This would be equivalent to achieving the 2.0 g/ test regulation set for 1982 cars. For the current average car the level will not significantly change from the 35 to 50% level until 4-5 years after 1982, as shown in Fig. B-26. This estimate indicates that the HC evaporative emissions level per the regulations could be within that allowed by 1987.

However, satisfying the regulation of 6 g/test and eventually 2 g/test in 1982 is not simply a matter of determining what the average car does. The important question is how significant and large are the variations among all cars, and does this sensitivity to the emission control configuration, fuel composition, and fuel vapor pressure preclude a prediction?

An example of the uncertainty in the data and the large variation depending on type of emission is shown in Fig. B-27. This figure shows the HC evaporative emissions in gm/shed test vs car mileage for diurnal, hot carburetor, and total evaporative emissions. The data points indicate the variability. The solid line on the diurnal curve displays the trend as a function of car mileage. In addition, data from the 1976 Los Angeles 20-car sample gives one estimate of diurnal emission in L.A. Note the large difference between the two mean diurnal curves. This is indicative of lack of correlation, large uncertainty in the sample, differences in cars and model years and differences in areas where the evaporation takes place.

Table B-1. Fuel Evaporative Emissions Using the Enclosure Technique (FY 75 Emission Factor Program)

1976 Model Year Cars in Denver and Los Angeles

		Diurnal Loss, gm		Hot Soak Loss, gm		Combined[a] Loss, gm/mi	
City	N	Mean	SD	Mean	SD	Mean	SD
Denver	20	21.74	13.06	10.46	6.20	1.91	0.93
Los Angeles	20	7.76	6.65	5.40	4.09	0.87	0.48

[a]Combined HC evaporative loss in gm/mi = diurnal loss + (No. trips per day x hot soak loss) divided by No. miles per day with 3.3 trips per day and 29.4 miles per day.

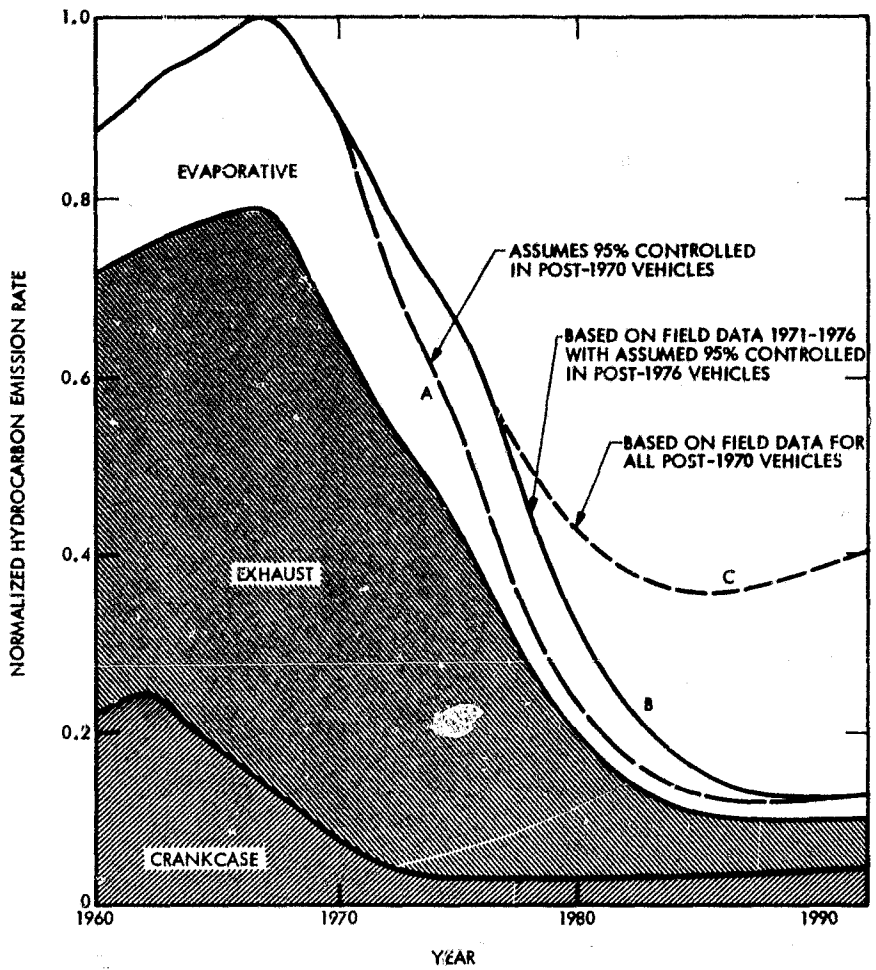


Fig. B-26. Normalized Urban Aggregate HC Emission Rate from In-Use Automobile Population (Ref. 2-1).

B.4 TRANSPORTATION NETWORK MODELS

Travel demand forecasting is a discipline that has been developed to evaluate the importance of transportation projects. These projects include short term projects such as controlling traffic lights or long term projects such as building new highways. Long range projects are primarily concerned with building new transportation facilities, whereas short term projects, also referred to as "transportation system management," are more concerned with achieving more efficient use of current transportation facilities. The outputs of the travel demand forecasting are primarily the forecasts of traffic volume and transit usage over the primary transportation network. Traffic volumes are generally measured in terms of Annual Average Daily Traffic (AADT). Forecasts of hourly traffic have been performed for many regions, including Denver and Los Angeles.

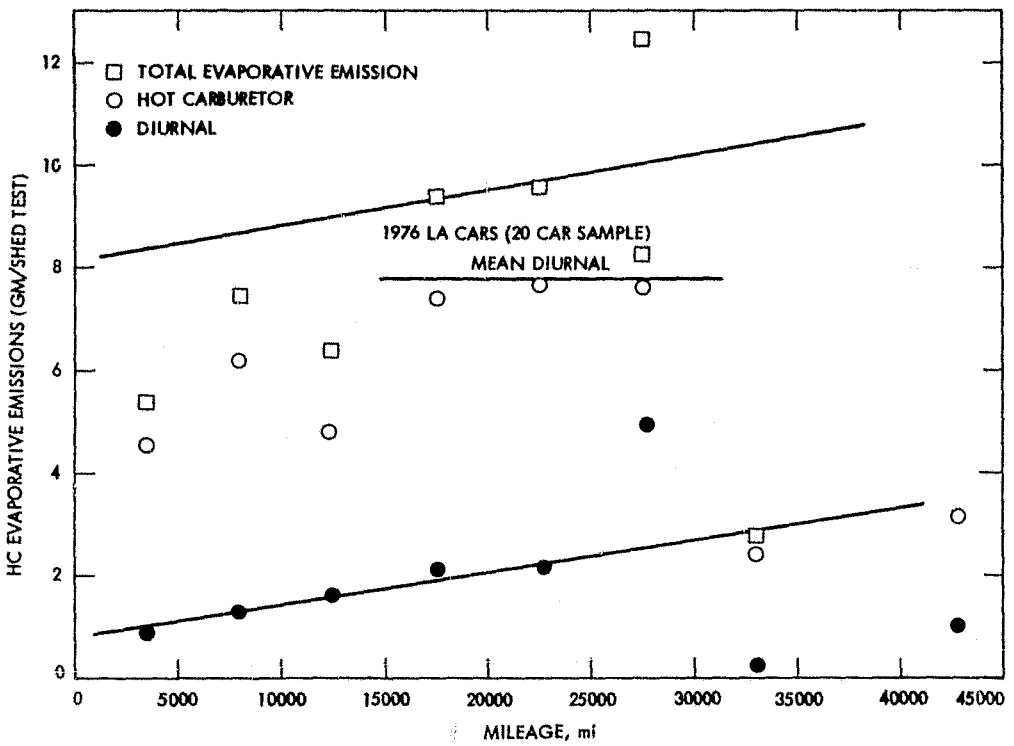


Fig. B-27. Hydrocarbon Evaporative Emissions versus Odometer Mileage

This section of the report describes a technique used to generate hourly estimates of the traffic over a day, which may be a function of the day of week and season. These temporal patterns, which are usually assumed to be same for each highway link, are combined with AADT to yield hourly traffic predictions over the traffic network.

Urban travel demand forecasting has been used recently to coordinate transportation planning and air quality management. Transportation strategies for improving air quality have been divided into two areas: vehicle emissions control and strategies relating to the usage of the automobile. These latter strategies relate to either reducing the usage of the automobiles or making more efficient use of the current transportation structure. Strategies aimed at reduced automobile usage include approaches such as transit improvements, carpool programs, and 4-day work week. Strategies relating to making more efficient usage of the current highway system include light synchronization, staggered work hours, widened freeways, and systems, reversing lanes, speed limit controls, ramp metering for freeways, and systems for handling emergencies more efficiently. The travel demand forecasting process requires the inputs indicated in Fig. 2-40. These will be described in the following sections.

B.4.1 Urban Activity

Urban activity is the forcing function behind trips to work, school, shopping, etc. The transit and highway networks shown in Fig. B-28 define the paths along which trips are constrained. In order to conveniently describe urban activities and the transportation networks, simplifications are introduced. First, the urban area is broken into "zones." These zones tend to coincide with standard census zones, which correspond to a population unit of about 5000. The physical size of zones tend to become smaller as population becomes more dense.

For example, Zonal Urban Activity Forecasts depend upon parameters as (Ref. B-15, p. 1-24):

(1) Total urban area population and employment estimates.

(2) Location and behavior of people and businesses.

(3) Local policies regarding land development, transportation, zoning, sewers, etc.

The purpose of urban travel demand forecasting is to relate travel activities to forecasts of zonal urban activities. For each zone, specific information must be collected. For example, for the LARTS system (Los Angeles Regional Transportation Study), the following zonal factors are collected (Ref. B-16):

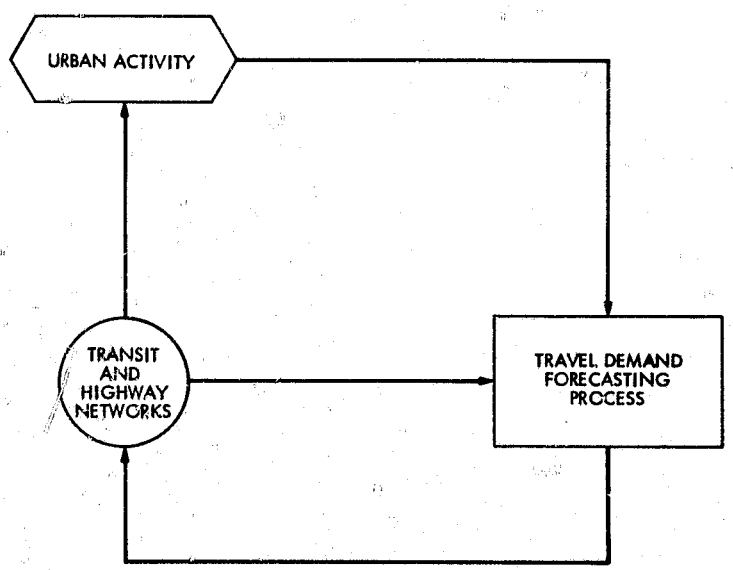


Fig. B-28. The Environment of Travel Demand Forecasting (Ref. B-15, p. 1-21)

(1) Population.

(2) Total employment.

(3) Retail employment.

(4) No. of single dwelling units.

(5) No. of multiple family dwelling units.

(6) Median household income.

These factors will be eventually related to forecasting number of trips, origin and destination for each trip, and the trajectory along which the trips are made. The transit and highway networks are described analytically to help predict travel patterns along major arteries. In some cases, minor arteries are included in an analysis. Collector street traffic is usually not predicted directly. These networks are specified in terms of highway links. Each link contains two nodes, which represent intersections of various highway or transit links. Other properties of these networks such as the level of service and the capacity may be used in determining the paths chosen between a specific origin and destination. Capacity is determined by the type of facility, speed limit, and the number of lanes. Level of service is related to the effective speed, which is also determined by the amount of congestion. Other parameters which can influence both the mode of transportation and the cost include parking availability and costs.

B.4.2 Travel Demand Forecasting

In order to simplify the conceptual process, one assumes that all traffic in a zone begins and ends at a single node, which is the centroid or "center of mass" of the population. This centroid is connected to the transportation network by a fictional network link, referred to as a "centroid connector."

Driving patterns of people are determined by Origin-Destination (O-D) surveys, which are sampled on a house-to-house basis to determine the following typical information (Ref. B-15, p. 2-15):
For each house:

(1) Address.

(2) Number of persons.

(3) Number of cars available.

(4) Occupation.

(5) Income.

For each trip:

(1) Who took trip.

(2) When trip began and ended.

(3) Times of beginning and end.

(4) Purpose at origin.

(5) Purpose at destination.

(6) Mode of transportation.

(7) Type of activity at origin and destination.

To determine patterns for trips that originate outside the planning region, roadway surveys are taken. Additional information that is used to develop models are truck and taxi surveys, counts of vehicle traffic and transit ridership. By comparing Figs. B-28 and B-29 it may be seen that urban travel demand forecasting involves the following four steps:

(1) Trip generation: Forecast number of trips to be made (trip ends).

(2) Trip distribution: Where trips go (connect ends).

(3) Mode usage: How trips will be taken by each mode of travel (car, transit).

(4) Trip assignment: Predicts routes trips will take, resulting in traffic forecasting for highway system and ridership forecasts for transit system.

These four steps are separately described in the following sections.

B.4.2.1 Trip Generation. Trip generation is the process by which trip ends are predicted for each zone. Each trip has two ends: a production end and an attraction end. All trips that begin or end at home are produced at home and attracted by the other end. Other types of trips are produced at the origin and are attracted by the destination. For the purpose of trip generation it is useful to separate out trips into different trip types, as classified by trip ends. For example, LARTS employs 5 different trip types (Ref. B-16, p. 5).

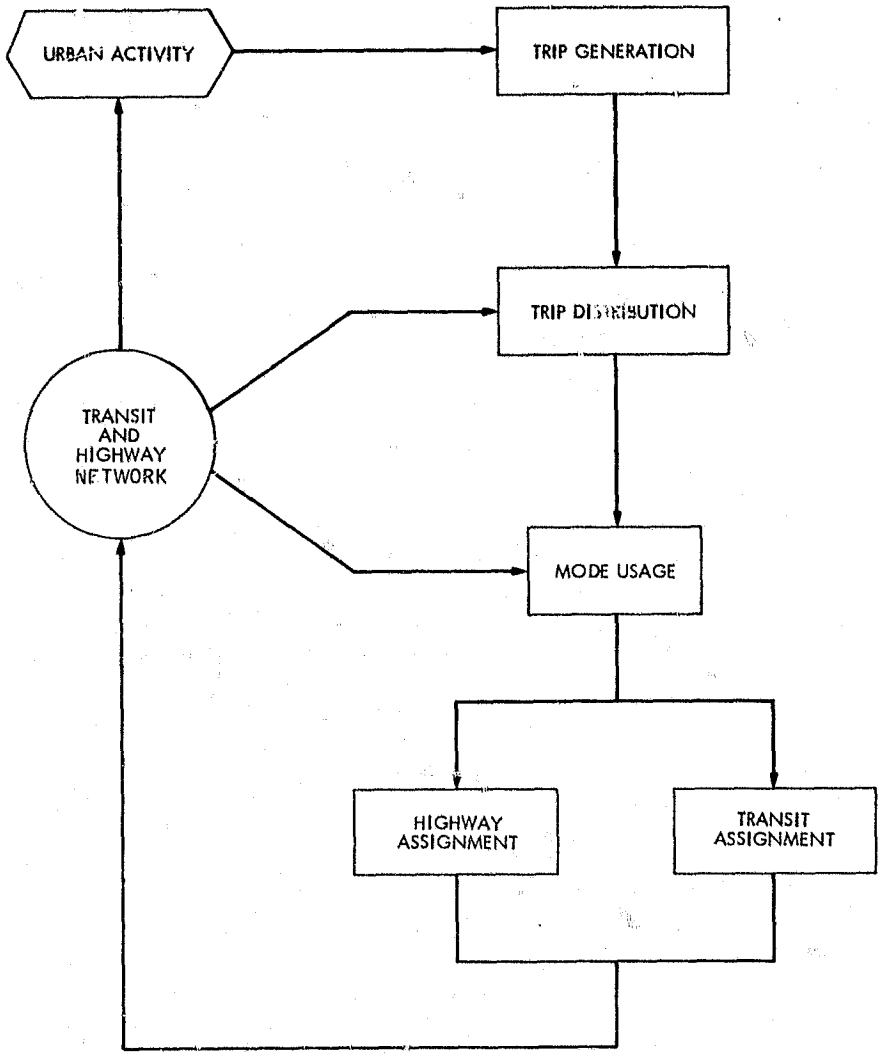


Fig. B-29. Structure of the Urban Demand Forecasting (Ref. B-15, p. 1-23)

(1) Home-Work

(2) Home-Shopping

(3) Home-Other

(4) Other-Other

(5) Other-Work

There are separate recommended models for different types of trip ends. Productions are handled separately from attractions.

The basic strategy of trip generation is to relate trips by specific purpose or trip type or to other variables relating to land use or economic activity. Several techniques such as multiple linear regression and cross classification are used to develop these models. Only the cross classification technique will be discussed. For example, trips generated at a household may be predicted on the basis of automobile availability and household income. Concerning household availability, households are divided into three categories: 0 cars, 1 car, 2+ cars. LARTS predicts the number of trips based on automobile availability, which is a function of type of housing unit, population per unit, ratio of single housing units to total units, median income, and country. Separate models are used to distribute total trips among the different trip types. Trips are attracted to various locations depending on the character, the location, and the activities taking place there. Work-related attractions are related to the number of employees. Shopping-related attractions are a function of shop square footage.

B.4.2.2 Trip Distribution. Since trip distribution is the process of matching trip ends, the output of this process is a trip table that relates trip production to trip attractions on a zone-by-zone basis. The most common methods employed for trip generation include the FRATAR method, the intervening opportunity method, and the gravity method. Only the last method will be discussed in this report.

The gravity model (Ref. B-15, p. 4-3) is summarized by the following equation.

$$T_{ij} = \frac{P_i A_j F(t)_{ij}}{\sum_{j=1}^{n} A_j F(t)_{ij}}$$

where:

T_{ij} = total number of trips produced in zone i and attracted to zone j

P_i = trips produced in zone i

A_j = trips attracted to zone j

$F(t)_{ij}$ = friction factors

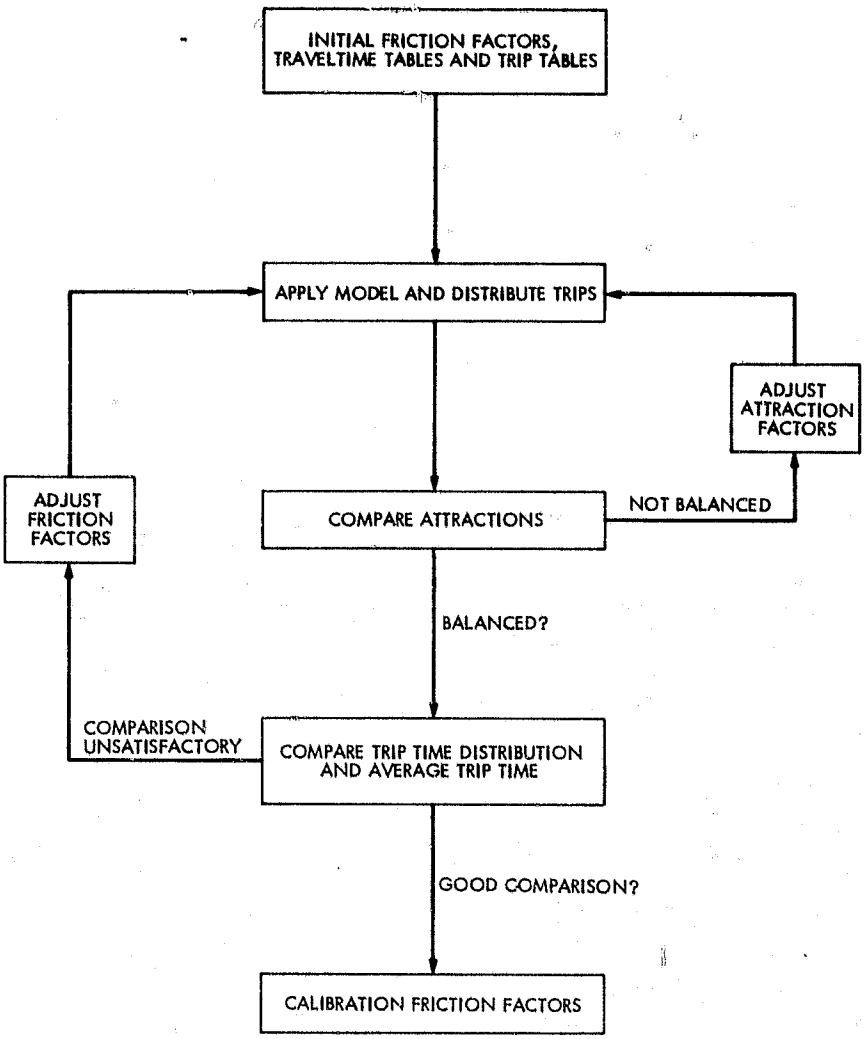


Fig. B-30. Calibration of the Gravity Model (Ref. B-15, p. 4-23)

The friction factor F_{ij} measures the tendency of people to want to travel from zone i to zone j. If the friction factor F_{ij} is large, then the propensity to travel from i to j is larger. Thus, "friction" factor is somewhat a misnomer. A better term would be "attraction factor." Friction factors are chosen to reflect the observed fact that people tend to prefer to minimize travel required to accomplish a given objective. Thus, people tend to go to shopping centers nearer their home, than to ones farther away. In some analyses F_{ij} is hypothesized to be inversely proportional to the travel time between zone i and zone j. Socioeconomic factors may also be used to cause predicted travel patterns to better agree with survey data.

In order to make the gravity model work, it is necessary to calibrate the model; i.e., determine the friction factors. In particular, although the gravity model automatically generates the appropriate number of attraction per zone, it does not necessarily generate the appropriate number of attractions (Ref. B-15, p. 4-22). Thus the process requires adjustment of the attractions to balance the process. Friction factors and socioeconomic factors must be empirically determined. Three items are used as input for the calibration process:

(1) Production-attraction tables for each trip purpose.

(2) Travel time (impedance) for all zone pairs.

(3) Initial friction factors.

Fig. B-30 summarizes the process for calibrating the gravity model.

B.4.2.3 Mode Usage. Mode usage models simulate personal travel choices involving alternative modes such as car, car pool, and public transportation. Factors that influence mode usage are (Ref. B-15, p. 5-3):

(1) Characteristics of tripmaker.

(2) Characteristics of the trip.

(3) Characteristics of the transportation system.

Some of the most important characteristics of the tripmaker that affect mode usage are (Ref. B-15, p. 5-3):

(1) Family income.

(2) Number of automobiles available.

(3) Education level.

(4) Family size.

(5) Family's age distribution.

(6) Type of dwelling.

(7) Residential density.

(8) Distance from tripmaker's dwelling to the central business district.

Characteristics that affect the choice of mode are trip type, trip distance, time of day, travel time and cost.

The mode usage model used by LARTS calculates a demand function only for home-work trips, which account for 45% of the trips reported in the 1967 home interviews. The survey showed that for every home-work transit trip, there was .09 other-work and 1.10 non-work transit trip. These factors are used to calculate these other trips based upon the home-work trips. The estimate of the home-work transit trips is based upon a two-step process:

(1) First, a marginal disutility function is calculated that presents the difference in "time" between use of the transit system and the automobile.

(2) The demand is calculated based upon empirical curves that relate demand to the disutility and the level of household income.

The marginal disutility function is as follows (Ref. B-16, p. 22):

$$U = (T_r - A_r) + 2.5\ (T_x - A_x) + \frac{T_c - A_o}{.25\ I}$$

where

U = the marginal disutility

T_r = transit running time

A_r = automobile running time

T_x = transit excess (access and waiting) time

A_x = auto excess (access and terminal) time

T_c = transit cost

A_o = perceived auto operation cost (include parking fee)

I = zonal median income

In this equation, excess time is seen to weigh a factor of 2.5 more than running time. The cost of time is calculated by dividing the costs by the tripmaker's perceived value of time, which has been shown to vary inversely with one quarter of his income.

For auto traffic, the mode use model must also specify the number of vehicle trips, which will be generally less than the person trips as more than one person can occupy a car. The factor that converts automobile trips to person trips is the "automobile occupancy factor." LARTS models this parameter as a function of trip distance.

B.4.2.4 Assignment. The object of the assignment phase is to determine which paths are chosen for the trips by both automobile and transit modes. In order to simplify the assignment process, all trips beginning or ending a zone end at a fictional point, "centroid," which corresponds to the center of mass of population. The centroid is connected to the network by a set of "centroid" connectors. Centroid connectors are artificial components of the network that represent all streets which are not specifically defined on the network.

Several techniques are available in determining the assignment of highway trips. One common approach (Ref. 2-21) begins by finding the path with the minimum impedance, measured in terms of cost, total time, or a combination of factors. The process of developing the minimum impedance paths between nodes is referred to as "path finding." The total process of developing the set of optimized paths is referred to as "path skimming."

The above process may lead to an unreasonable assignment of traffic volume on specific links (Ref. B-15, p. 6-16). Several techniques are used to modify the assignment process to obtain more realistic estimates.

(1) The relationship between speed and traffic volume is used to modify the link speeds; the link loading process is repeated 3 or 4 times until a balance is obtained between speed, volume, and capacity.

(2) Trips are assigned to several paths between zones, the distribution depending upon the distribution of impedance among the most probably paths.

Employing a variation of both of the above techniques, LARTS assigns the trips in two steps. First, work trips are assigned using a set of low speeds for the network representing peak traffic condition. Second, other trips are assigned using a set of high speeds for off-peak traffic conditions. The traffic is split 50-50 (Ref. B-16, p. 26) between a minimum time path and a minimum city street path. Person home-work trips allocated to transit are assigned to minimum time paths using peak traffic assumptions.

REFERENCES

B-1 Milks, D., and Matula, R.A., "Emissions and Fuel Economy Test Methods and Procedures For Light Duty Motor Vehicles--A Critique," SAE 760141, Mechanical Engineering and Mechanics Dept., Drexel University, presented at the Automotive Engineering Congress and Exposition, Detroit, Michigan, February 23-27, 1976.

B-2 Midvrsi, T.P., and Castaline, A.H., Determination of Percentages of Vehicles Operating in the Cold Start Mode, EPA-450/3-77-023, GCA Corporation, Burlington Road, Bedford, Massachusetts 01730, August 1977.

B-3 Moore, L., and Unger, D.C., Jr., A Study of Exhaust Emissions from 1975-1979 Model Year Passenger Cars in Los Angeles, Prepared for EPA, Environmental Systems, Inc., 7300 Bolsa Ave., Westminister, CA 92683, July 1979.

B-4 Dohner, A.R., "Transient System Optimization of an Experimental Engine Control System over the Federal Emissions Driving Schedule," SAE 780286, General Motors Research Laboratory, 1979.

B-5 Juneja, W., et al., "A Treatise on Exhaust Emission Test Variability," SAE 770136, Vehicle Emission Laboratories, General Motors Corporation, presented at the International Automotive Engineering Congress and Exposition, Cobo Hall, Detroit, February 28-March 4, 1977.

B-6 Mobile Source Emission Factors, Final Document, PB 295672, EPA-400/9-78-005, US/EPA, Office of Air and Waste Management, Washington, DC 20460, March 1978.

B-7 Cross, R.H., "Emissions from Catalyst Equipped California Cars in Customer Service," SAE 800395, California Air Resources Board, presented at Congress and Exposition Cobo Hall, Detroit, Michigan, February 25-29, 1980.

B-8 Walsh, M.P., and Nussbaum, B.D., "Emissions from Catalyst Cars Beyond 50,000 Miles and the Implications for the Federal Motor Vehicle Control Program," SAE 780027, presented at Congress and Exposition, Cobo Hall, Detroit, Michigan, February 27-March 3, 1978.

B-9 Evans, L., "Urban Traffic, Fuel Economy and Emissions--Consistency of Various Measurements," SAE 780934, Traffic Science Dept., General Motors Research Labs., Warren Michigan, 1979.

B-10 Compilation of Air Pollutant Emission Factors, Third Edition (Including Supplements 1-7), AP-42 Part A, US/EPA Office of Air and Waste Management, Office of Air Quality Planning and Standard, Research Triangle Park, N.C. 27711, August 1977.

B-11 Cantwell, E.N., Jr., et al., "Projections of Motor Vehicle Fuel Demand and Emissions," Petroleum Laboratory, E.I. DuPont De Nemours and Co. (Inc.). SAE 780933, 1979.

B-12 Ostrouchov, N., "Effect of Cold Weather on Motor Vehicle Emissions and Fuel Economy," SAE 780084, Mobile Sources Div., Fisheries and Environment, Canada, presented at Congress and Exposition Cobo Hall, Detroit, Michigan, February 27-March 3, 1978.

B-13 Scubar, R.L., et al., "Soak Time Effects on Car Emissions and Fuel Economy," SAE 780083, Southwest Research Institute and US/EPA, presented at Congress and Exposition Cobo Hall, Detroit, Michigan, February 27-March 3, 1978.

B-14 Rutherford, J.A., Automobile Exhaust Emission Surveillance-Analysis of the FY 1975 Program, EPA-460/3-77-022, US/EPA, Emission Control Technology Division, Ann Arbor, Michigan 48105, December 1977.

B-15 "An Introduction to Urban Travel Demand Forecasting," Federal Highway Administration Urban Mass Transportation Administration, 1977.

B-16 Sorenson, S., and Brewer, D., "The LARTS Transportation Model: Description and Assumptions," Technical Report Series TR/2, CALTRANS.